MAPPING ELK DISTRIBUTION ON THE CANADIAN PRAIRIES: APPLYING LOCAL KNOWLEDGE TO SUPPORT CONSERVATION

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By

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ABSTRACT

Once abundant across the Great Plains of North America, prairie-parkland elk (*Cervus canadensis manitobensis*) underwent a catastrophic population collapse and dramatic contraction of their overall range through the late 1800's and the early 1900's due to habitat loss (primarily from agricultural expansion) and unregulated hunting. Populations were able to recover in some areas following new hunting regulations and the establishment of protected areas. Prior to this study, the current distribution of prairie parkland elk was poorly understood, though it was established that they were largely relegated to large protected areas and made use of adjacent agriculture dominated landscapes. In order to effectively manage prairie-parkland elk so populations remain resilient to ongoing habitat loss, population reduction and disease risks, detailed mapping of their range and an understanding of the environmental factors most important to elk is essential. The purpose of my thesis is to characterize elk distribution and resource selection patterns in the prairie-parkland of Saskatchewan and Manitoba, Canada, at a landscape scale and assess the implications of distribution for species persistence, by using both local ecological knowledge (LEK) and biological research techniques in tandem.

Integrating LEK with more conventional biological research can provide complementary data at contrasting time and spatial scales and facilitates comparison of multiple independent datasets. Furthermore, LEK research creates important opportunities to engage stakeholders in contributing knowledge and may facilitate relationships and contribute toward more effective resource management. I used three sets of biotelemetry-collar data from across Saskatchewan and Manitoba (*n* = 328 collared elk; 1998–2012), in conjunction with LEK from hunters, biologists and enforcement officers (*n* = 71 participants) to create a series of resource selection functions (RSFs) characterizing elk distribution across Saskatchewan and Manitoba. I gathered LEK in workshops where participatory mapping was conducted with local experts across the study area. The RSF analysis determined that prairie-parkland elk selected locations close to protected areas and far from high road density. Elk also selected areas with moderate amounts of mixed-wood and deciduous forests and herbaceous vegetation. Models developed with LEK only, biotelemetry collar data only and a combined dataset were all validated against an independent dataset of elk crop damage locations. All models predicted elk presence well. The RSF scores of the LEK only and radio-collar only models were not significantly different.

Successful conservation management requires identifying which areas are most important to a species, and assessing if these areas are vulnerable to threats, as well as balancing human resource needs. Using the RSF-based maps of prairie-parkland elk distribution, I identified locations of high quality habitat (top 10% of RSF values) and determined which of those areas were vulnerable to agricultural expansion, forestry, disease, and hunting. I identified 81 high quality habitat areas with a combined total area of 30 753 km^2. One or more vulnerability factor impacted 87% of the identified core area. High quality areas were clearly clustered around the boreal-prairie transition zone and large protected areas. The majority (88%) of high quality core areas were located within protected areas. A connectivity analysis using least cost path analysis determined that core habitat areas endemic with chronic wasting disease (CWD) in wild elk are highly connected to other high quality habitat areas. CWD thus has the potential to reduce cervid populations within the study area.

My thesis results highlight that prairie-parkland elk populations in Saskatchewan and Manitoba remain threatened by habitat loss and disease, and emphasize the need for habitat conservation to prevent further population reduction. While elk populations have regained a small fraction of the range lost at the turn of the last century, they have not been able to re-establish with much success in the prairie portion of their range. I also determined that LEK can be as effective as conventional biological research approaches to develop RSFs. I also confirmed that stakeholders within the study are knowledgeable about elk behaviour. The distribution maps and identified areas of priority concern created in this thesis can provide important insights to support the management and maintenance of abundant elk populations.

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LIST OF ABBREVIATIONS

AIC – Akaike's information criterion

ΔAICc – Difference in Akaike's information criterion adjusted for small sample size

Bovine TB – Bovine tuberculosis

CWD – Chronic wasting disease

ENFA – Ecological niche factor analysis

GIS – Geographic Information System

LEK – Local ecological knowledge

PGIS – Participatory GIS

RSF – Resource selection function

SDM – Species distribution model

TEK – Traditional ecological knowledge

CHAPTER 1: GENERAL INTRODUCTION

1.1 Introduction

"We can't solve problems by using the same kind of thinking we used when we created them."
- Albert Einstein

Effectively addressing the conservation issues that define current wildlife management requires more than just an understanding of the biology of a species (Folke 2004; Reed 2008). Human factors, such as land use practices, economic influences, and beliefs also significantly influence the success of conservation initiatives (Clark 2002). Conventional problem solving strategies historically taught to university-trained biologists are based in rationality, and depend on specific problem definition and directed data collection (Berkes 2004; Watson & Huntington 2008). As such, when conducting research to address these complex socio-ecological problems, biologists typically focus on collecting short-term, biological data. These strategies are rarely adequate to solve problems that are typically large in scale and scope, uncertain, and changeable and have both biological and social aspects (Kates 2001).

Conserving species while balancing necessary human activities such as agriculture can be called a “wicked problem”, which is defined as a problem with “no definitive formulation, no stopping rule, and no test for a solution” (Rittel & Webber 1973; Hisschemöller & Hoppe 1995; Ludwig 2001; Berkes 2004). In these situations, there is no single correct answer. Yet despite these challenges, decisions must be made about resource management and species conservation. Consequently, it is imperative that conservation efforts look beyond conventional biological research tools and access all knowledge systems at their disposal. Collection and application of local ecological knowledge is one approach that can be used to span disciplinary and cultural gaps, and bring research toward an alternative management ideology grounded in the inclusion of multiple perspectives and value systems (Roling & Jiggins 1998; Folke 2004; Berkes 2004).

Local ecological knowledge (LEK) can be defined as the individual insights derived from personal observations and experience that occur when living and working in the natural environment (Gilchrist et al. 2005; Brook & McLachlan 2008). In contrast, science is defined as “the pursuit and application of knowledge and understanding of the natural and social world following a systematic methodology based on evidence” (Science Council 2009). As such, there are important differences and similarities between LEK and ‘Western’ or ‘expert-based’ science.

LEK is valuable in its own right to understand social-ecological systems, and can be used to develop hypotheses, and interpret science-based results (Riedlinger & Berkes 2001; Moller et al. 2004). LEK is inherently different from information obtained from conventional biological research techniques, although no less valuable (Briggs 2005). For instance, LEK often differs in scale, both temporally and spatially, from data gathered in technical biological research (Gagnon & Berteaux 2009). Researchers are increasingly using LEK in biological studies to empower and engage communities, as well as gather types of data that would otherwise be financially or logistically impossible (Davis & Wagner 2003; Brook & McLachlan 2008; Anadón et al. 2009). At a fundamental level, the inclusion of knowledge from individuals other than biologists and researchers is an important step in recognising the utility of other ways of knowing and different ways of understanding the environment (Agrawal 1995). Employing LEK as a research technique has become a method to involve people in the difficult and value laden process of wildlife management.

Elk (*Cervus canadensis*) are a particularly complex species for wildlife managers and conservationists as they can be thought of both positively, due to associations with hunting, ecotourism and wildlife viewing, and also negatively, as a result of interactions that have large socio-economic impacts such crop damage, disease transmission to livestock and highway collisions (Wisdom & Cook 2000). Elk are a frequent source of human-wildlife conflict, yet many people also value elk as a food source and a symbol of wilderness (Brook 2009). Whether people enjoy elk on the landscape or despise them is highly variable and context dependent. Opinions on elk presence can differ based on the livelihood of the individual or the ecosystem type they live in. Most often opinions are determined by the direct effects of elk in the area where the individual resides (Wisdom & Cook 2000; Brook & McLachlan 2006; Brook 2009). Elk are a keystone species, and consequently the ecological impact of elk presence in an area is significant. Elk alter the ecosystem they inhabit by serving as a key food source for wolves, and by modifying the successional trajectories of plant communities (Carbyn 1983; Ripple & Larsen 2000; Kie et al. 2003). Management or conservation initiatives that include elk need to incorporate the perspectives held by those who are impacted by elk and interact with them regularly, in addition to those who understand the overreaching ecological impacts of elk. Therefore, balancing conservation of elk and their habitats with the need to maintain land use practices such as forestry and agricultural productivity could be thought of as a wicked problem.

Prior to European settlement, elk were the most abundant cervid in North America, with estimated populations numbering close to 10 000 000 (Bryant & Maser 1982) . Settlement of the prairies and parkland led to many changes, including increased hunting pressure, dramatic habitat modification converting native grasslands to farmland, and alterations to the fire regime. The combination of these factors resulted in a drastic range and population collapse for prairie elk (Bryant & Maser 1982; Wisdom & Cook 2000; Brook 2009). In the early 1900s, populations in the Canadian prairies were reduced to a small number of isolated pockets in strongholds along the boreal forest-prairie interface (Bryant & Maser 1982). Elk numbers have since increased above those historic lows of approximately 600; the current Saskatchewan elk population is around 15 000, while Manitoba elk number roughly 7350, although most elk populations within the study area are not closely or routinely monitored. Elk appear to remain closely tied to a small number of large protected areas, and have not recovered most of their former range (Polziehn et al. 2000; Laliberte & Ripple 2004; Arsenault 2008; Brook 2010; Manitoba Conservation 2013). Prior to my study, a detailed understanding of contemporary elk distribution in the prairie provinces was lacking, despite the fact that this knowledge is crucial to accurately assess the potential for species persistence in the region.

When managing a species with the goal of long term persistence, it is critical to understand the species' geographical distribution and habitat selection (Pearce & Boyce 2006; Austin 2007). Several presence-only modeling techniques can be used to characterize the distribution of a species (Guisan & Zimmermann 2000; Pearce & Boyce 2006). These modelling techniques generally focus on quantifying animal-environment interactions and use the measurement of this interaction to predict distribution at local, landscape, and regional scales (Guisan & Thuiller 2005; Boyce 2006). A Resource Selection Function (RSF) is a method commonly used to characterize and map resource selection (Johnson & Gillingham 2008; Rice et al. 2013). RSFs determine how animals choose to meet their basic requirements for reproduction and survival by identifying the probability of use of a resource unit by the study animal or population (Manly et al. 2002; Boyce 2006). A resource unit is defined as a biotic or abiotic factor that impacts an animal's ability to live and reproduce (Thomas & Taylor 2006). An RSF is built on the reasonable assumption that animals will disproportionately select resources relative to the availability of that resource, if that resource affects the animal's fitness (Thomas & Taylor 2006). RSFs can be valuable stand-alone tools used to quantify the relationships between animals and specific resources, facilitate the mapping of species distribution, and as building blocks for additional modeling techniques (Chetkiewicz & Boyce 2009).

To ensure robust elk populations in Saskatchewan and Manitoba, currently unknown information regarding the location of critical elk habitat, and threats to that habitat need to be clarified (Cianfrani et al. 2010; Hebblewhite et al. 2012). There are many methods used to identify specific areas important to species conservation (Wilson et al. 2005). One approach by Pressey and Taffs (2001), focuses on measures of vulnerability and irreplaceability of an area. Vulnerability is explained as the risk that an area that is part of species habitat will be modified or harmed by extractive industries, human use, or biological occurrence such as disease (Margules & Pressey 2000). The irreplaceability of an area is the value that area has as wildlife habitat, or in other words, the contribution it makes towards a conservation goal (Pressey & Taffs 2001; Carroll et al. 2003). The predictive values of an RSF can serve as a measure of irreplaceability (Carroll et al. 2003). While solely biological approaches have proven invaluable for describing and identifying critical habitat, an important limitation is that these methods lack explicit opportunities to engage communities and are data intensive in that they require large volumes of data, which are expensive to obtain and not always available. Both LEK and conventional scientific data are valuable in their own right and can be used individually to address conservation goals, but together may provide especially powerful opportunities for understanding and managing socio-ecological systems.

1.2 Thesis Objectives

To solve the multifaceted conservation problems of today, biologists can benefit from using a broader range of tools. LEK is infrequently used in conjunction with spatial biological data, even though it has the potential to greatly add to the depth of knowledge available for analysis and encourages interdisciplinary thinking (Balram et al. 2004; Anadón et al. 2009; Brook & McLachlan 2009). The purpose of my thesis was to characterize elk distribution and resource selection patterns in the prairie-parkland of Saskatchewan and Manitoba, Canada at a landscape scale, then ascertain the implications of the distribution on population persistence by using local ecological knowledge (LEK) and biological research techniques in tandem. My specific objectives were to: (1) generate spatial models of elk distribution using three different elk location datasets; (2) validate and compare the distribution models produced; (3) identify core areas of high quality elk habitat and assess the vulnerability of these areas to habitat loss, disease and population reduction; (4) clarify the role protected areas play in determining high quality elk habitat; and (5) define areas of priority conservation concern for elk.

1.3 Hypotheses

In this thesis, I set out to determine elk distribution in the prairie-parkland region of Saskatchewan and Manitoba. If elk distribution is determined by a combination of environmental covariates, then I will be able to predict where elk are located in the study area. I predict that elk will select areas with intermediate cover (Wisdom et al. 1986; Boyce et al. 2003), are far away from high traffic roads (Lyon 1979, 1983; Unsworth et al. 1998) and provide access to quality forage. I also predict that core areas for elk populations will be based along the transition zone between boreal forest and agriculture, as elk favour heterogeneous habitats with access to cover and open land to graze on (Lyon 1979; Wisdom et al. 1986) .

For my hypotheses, I am following Chamberlin's method (1890) of multiple competing hypotheses. He advocated for developing a series of multiple hypotheses *a priori*, instead of the traditional null and alternative hypothesis format that is commonly used in scientific studies. This approach allows a set of cogent explanations to be fully explored, and prevents bias of the researcher towards one possible result, thus skewing the interpretation of the results (Chamberlin 1890). Chamberlin's method is commonly used in ecological modelling and is an appropriate choice for studies that lack a true experiment, such as those examining resource selection by free-roaming wildlife (Dochtermann & Jenkins 2010). Specifically in the case of an RSF, developing a set of *a priori* hypotheses is extremely valuable, as the statistical significance alone of individual predictor variables is rarely insightful, can identify spurious relationships, and fails to account for important ecological interactions among variables (Boyce et al. 2002). RSFs are built on the assumption that animals do preferentially select resources, which is a characteristic of all organisms, thus testing the statistical significance of these results would only indicate whether this assumption was met (Cherry 1998; Manly et al. 2002).

Based on this philosophy, I developed 10 *a priori* models to predict elk distribution with 15 environmental predictor variables. (See Table 3.2 on page 44) These variables were chosen based on a survey of previous elk habitat selection studies.

1.4 Thesis Structure

This document follows the format of a manuscript style thesis as outlined by the University of Saskatchewan College of Graduate Studies and Research. In Chapter 1, I provide a general overview on the background and purpose of the thesis. Chapter 2 reviews the relevant literature. Chapters 3 and 4 are original data-based research chapters and are presented in a standalone manuscript format. In Chapter 5, I conclude the thesis by providing a summary and brief discussion of the results presented in the document. All citations and references in this thesis follow the format outlined by the journal Conservation Biology.

CHAPTER 2: LITERATURE REVIEW

2.1. Conservation biology and human wildlife conflict

Worldwide, many wildlife populations are declining. The global extinction rate currently exceeds estimated pre-human extinction rates by 100–1000 times (Pimm et al. 1995). Conservation biology aims to protect organism communities and populations from extinction (Soulé 1985), with the ultimate goal of maintaining or restoring biodiversity (Lindenmayer & Hunter 2010). What differentiates conservation biology from other biological fields is its crisis driven nature. In conservation biology, decisions and action are often required before all of the facts regarding a situation are known (Soulé 1985). While the reasons for the increasing extinction rate and number of species at risk in North America are multifactorial, including over exploitation, introduced species, pollution, and natural disasters (Kerr & Cihlar 2004; Venter et al. 2006), the largest driver worldwide is habitat loss (Schipper et al. 2008).

Extinction vortices are environmental changes that result in positive feedback loops which amplify the extinction risk of a vulnerable species (Gilpin & Soulé 1986). Events that cause extinction vortices include demographic variability (random variation in birth and death rates); reduced genetic heterozygosity leading to loss of fitness; decreased genetic diversity resulting in diminished resilience and ability to recover from stochastic events; and habitat fragmentation (Gilpin & Soulé 1986). Habitat fragmentation is a landscape-level process that occurs when habitat is broken up and the landscape is modified from its natural state. Clear cuts, road creation and agriculture development are examples of activities that can contribute to fragmentation (Fahrig 2003). The concept of habitat fragmentation originated in island biogeography theory (MacArthur & Wilson 1967). Initial research suggested that the effects of fragmentation on populations are twofold: first, there is the effect of the loss of habitat and secondly, the effect of the isolation of those patches from each other (Wilcox 1980). However, more recent research argues that the habitat loss component of fragmentation causes more detrimental effects to a population or species than the isolation created from breaking up habitat blocks (Debinski & Holt 2000; Fahrig 2003). Although the impact of fragmentation depends on habitat patch size, proximity to other patches and species' ecology, habitat alteration undoubtedly makes species more at risk of extinction and future population decline (Ewers & Didham 2007; Collins & Kays 2011). While these impacts are largely negative, there may also be some benefits of fragmentation, for instance in the control of disease spread (Dugal et al. 2013)

Fragmentation is chiefly caused by anthropogenic land use change (Haberl et al. 2007). The process of harvesting natural resources, through either extractive industries such as mining, or renewable processes like forestry (Rudel et al. 2009), and land conversion for direct human use through agricultural, construction of roads, highways, power lines, and residential developments (Vitousek et al. 1997) all modify and fragment habitat. As landscape modification intensifies, so does fragmentation. Therefore, understanding how wildlife interacts with a heterogeneous and anthropogenic dominated landscape is an essential component of conservation biology (Wiens et al. 1993; Forester et al. 2007). In addition to biological dynamics, socio-economic aspects that influence species persistence must also be considered.

Human-wildlife conflict occurs when the needs and behaviour of wildlife negatively impact the goals of humans, and vice versa (Madden 2004). Conflict between humans in rural areas and large herbivores has been a common occurrence across the world and throughout history (Treves et al. 2006). Human-wildlife conflicts can result in important human-human conflicts, further complicating conservation initiatives (Brook 2009). When domestication of livestock occurred, the role of wild herbivores changed from primarily that of a food source to a competitor for land and food resources needed by their livestock (Gordon 2009). In addition, encounters between wildlife and agricultural producers have shifted from being relatively simple conflicts between those directly impacted, to a social-political issue, entangled with broader concerns such as the exclusion of indigenous people from protected areas and endangered species management (Treves et al. 2006). These changing attitudes, combined with increasing agricultural expansion and habitat loss, have led to extensive and increasing human wildlife conflict in both developed and developing countries (Messmer 2000). For herbivore-human conflict in North America, conflict is generally caused by crop damage incidents (VerCauteren et al. 2006; Gooding & Brook 2014) but more recently, wildlife disease has become an important point of dramatic conflict.

Although damage by small pests such as insects or mice often causes the greatest cumulative crop loss over many small depredation incidents, people typically perceive large-scale damage events as more costly. Larger animals are therefore more vulnerable to the negative fallout that often occurs following damage (Treves et al. 2006). Due to their size and diet, large cervids in particular are prone to human-wildlife conflict (Gordon 2009). Livestock and ungulates evolved sympatrically, which resulted in overlapping ecological niches and direct competition for food resources (Wisdom & Thomas 1996; Gordon 2009; Sorensen et al. 2014). Additionally, due to their evolutionary closeness, ungulates, including cervids, and livestock can often pass disease back and forth as well as infecting humans. In this way, wildlife can serve as a reservoir for the disease and can re-infect livestock (Hudson et al. 2002; Lees 2004). In the Riding Mountain area of Manitoba, this reservoir-reinfection dynamic is partly responsible for a history of extensive human-wildlife conflict between farmers and elk (Brook 2009).

As is the case around Riding Mountain National Park, protected areas and neighbouring farmland are often ground zero for human-wildlife conflict (Madden 2004; Naughton Treves 2008). The intensification of conflict in these areas is caused in part by agricultural and the supporting residential developments modifying and disrupting key wildlife habitat. These modifications result in an increased concentration of wildlife populations in the areas that remain available for them, typically pastures or fields of crops. In such cases, human-wildlife conflict may become common (Messmer 2000). For instance, prior to European settlement, elk across North America moved seasonally between high altitude pastures in the summer, and low altitude, protected ranges in the winter. However, winter ranges are also desirable from a human perspective and many have since been converted to agriculture or developed as residential areas (Vavra 2006). Elk living in these landscapes survive in “ecologically incomplete systems”, where the remaining natural landscape available to them cannot support their year round resource requirements (Cole 1971). The lack of available nutritional resources can cause elk to move into agricultural lands in the winter to access the resources necessary for their survival. The extent to which elk depend on crops as a winter food source is situation dependent but can progress in extensively developed areas to the point where crops are a primary food source (VerCauteren et al. 2006). Such consumption of crops can cause considerable stress and negative socio-economic impacts on farmers (Walter et al., 2010).

A principal tenet of conservation biology is that human values shape conservation efforts (Lindenmayer & Hunter 2010). Many people living in a rural area have a different relationship with the environment than urban dwellers, in part due to a direct dependency on the natural world for their livelihood (Davis & Wagner 2003). In the case of human-wildlife conflict, how individuals are personally impacted by wildlife, in part, determines how they view the wildlife

(Messmer 2000). Thus, any conservation effort that involves wildlife prone to conflict, such as cervids in agricultural areas, should include a variety of perspectives and reflect the needs and desires of those living most closely with wildlife, while remaining based on appropriate conservation-based policies.

2.2. Local and traditional ecological knowledge in biological research and wildlife management

"*Knowledge freely available to all does not benefit all equally*." - (Agrawal 1995)

In the last two decades, the inclusion of traditional, local and indigenous ecological knowledge in biological research has increased (Davis & Wagner 2003; Brook & McLachlan 2008). Traditional ecological knowledge (TEK) or indigenous knowledge can be defined as "all types of knowledge about the environment derived from experiences and traditions of a particular group of people" (Usher 2000). Local ecological knowledge (LEK) can be thought of as the individual insights derived from personal observations and experience that occur when living and working in the natural environment (Gilchrist et al. 2005; Brook & McLachlan 2008). Both LEK and TEK can be used to identify hypotheses, provide information to researchers and dictate management strategies (Riedlinger & Berkes 2001), although TEK is often considered to be more contextual. TEK is deemed to be part of a broader cultural worldview that includes belief and spirituality, in addition to observation (Houde 2007); however, in practice all knowledge forms are somewhat rooted in context , individual perspective and culture. TEK is typically part of an ontology different from Western society (Simpson 2001). Observations or facts in TEK cannot be separated from the broader context of the epistemology, without fundamentally modifying the meaning of the knowledge (McGregor 2000). Although TEK and LEK are different, they are often used interchangeably in the literature (Agrawal 1995; Steele & Shackleton 2010). To summarize, TEK is knowledge held by a culture of people and is heavily influenced and dictated by their worldview, while LEK, although still value based, is knowledge held by individuals or small groups of individuals derived from direct personal experience in the environment (Huntington 2000).

The use of TEK and LEK has arisen from the realization that local people who live on the land are effective at providing valuable biological and social insight relevant to a problem or research question (Huntington 2000; Folke 2004). Use of LEK and TEK is also driven by the need to characterize complex systems, limited resources available for research and monitoring, and the urgency associated with conservation decision making (Martin et al. 2012). TEK has been recognized as both complementary and equal in importance to scientific knowledge (United Nations Environment Programme 1998; Berkes et al. 2000a). The increase in the recognition of the value of TEK and LEK reflects a changing lens for viewing ecological problems. Many conservation scientists have now acknowledged that environmental initiatives and issues exist in tandem with social factors. That is, society and groups of individuals can influence the outcome of conservation initiatives (Berkes 2004). People are a part of the ecosystem, not separate from it (Roling & Jiggins 1998). This idea exists in contrast to traditional Western wildlife management philosophy, such as the North American Model of Wildlife Management, where management policies are created based on two key factors: the sufficiency of biology as a problem solving tool and the overwhelming influence of expert authority and biological data. As a result, in this management paradigm, biologists are the ones making all decisions (Riley et al. 2002). Within these systems, alternative approaches that integrate LEK are TEK are often dismissed or relegated to second class data (eg: Gilchrist et al. 2005).

It has become apparent that Western resource management has often not resulted in environmental or ecological sustainability, thus a new research and management paradigm may be desirable (Davis & Ruddle 2010). Indigenous people have often managed their own resources successfully for many generations by employing a modified form of adaptive management. Adaptive management is the process of implementing a management decision as an experiment. The outcome of the decision is assessed and the action taken is then modified to improve the outcome. This is an ongoing process of reflection and adaptation, and is essentially learning by doing (Walters & Holling 1990). Indigenous management approaches are also based on learning by doing, and have the potential to provide insight into current problems (Berkes et al. 2000a). Using TEK or LEK in research projects can provide data that differs from data collected using conventional expert-based research methods, often with a different and complementary depth of knowledge. Length, spatial scale, and budget and overall resource issues often restrict what is feasible in research, including TEK or LEK in the research project is one way to expand research scale and minimize costs (Brook & McLachlan 2008). Observations from TEK and LEK generally span a long time scale with a highly local and specific spatial scale (Gagnon & Berteaux 2009). Although observations from TEK or LEK can be imprecise and qualitative, the time scale associated with both local and cultural observations, as well as the depth of highly localised knowledge, is valuable and a unique contribution to research (Moller et al. 2004). Including TEK or LEK in biological management has value as a tool to provide additional insight, but also promotes the use of a cultural framework of respect and reciprocity when doing research with communities (Kimmerer 2002). In contrast to Western practices where people are in control of the natural world, TEK promotes a holistic framework where connectedness or relatedness to the natural world is promoted and non-human organisms are treated respectfully (Pierotti & Wildcat 2000). In addition, including local community members in natural resource management initiatives is more successful and sustainable in the long term than relying on contracted outside assistance, and can result in community empowerment (Danielsen et al. 2009).

While some researchers believe that any inclusion of TEK or LEK in a research project is a positive step towards the addition of other perspectives in biological research, others argue that the value-laden nature of TEK makes the process fundamentally problematic (Nadasdy 1999). Many criticisms arise from the fact that most researchers believe science to be truly objective and separate from value-based, objective judgements (Brook & McLachlan 2005). On the contrary, research is a fundamentally value-driven and subjective process, and the worldview and biases held by the researcher can have a profound impact on how the data is collected, processed and interpreted. Nadasdy (2003) argues that the simple act of translating TEK into data compatible with wildlife management has the potential to alter the content and meaning of the knowledge, and is a fundamentally political process impacted by power dynamics. Additionally, he argues that while the scientific community uses examples of successful TEK integration within biological research, success often means that only TEK components that are validated by, or support the conclusions found by scientific research, are included. That is, the data collected and conclusions made by conventional research methods holds a trump card to alternative approaches or conclusions. Another facet of this concern is that TEK is often being defined and constructed by researchers outside of indigenous communities (McGregor 2000).These researchers are often only interested in the aspects of knowledge that may provide insight into ecological issues and ignores more complex aspects such as spirituality (Simpson 2001).

Solutions for successful integration from a research perspective vary. In the literature, approaches for successful integration are focused on strategies for dealing with differences in the perceived degree of accuracy between knowledges. Some argue that the emphasis on separating TEK/LEK from conventional biological research further promotes hierarchy of one knowledge type over the other (Agrawal 1995), and that separating knowledge systems promotes the belief that Western knowledge is grounded in rationality, where TEK embodies emotion and spiritual beliefs (Briggs 2005). Some believe that the reliability of LEK should be validated against biological research data to prove its accuracy (Gilchrist et al. 2005; Anadón et al. 2009), while others believe fully separating LEK from biological research and comparing results to other local knowledge, will ensure the validity of TEK/LEK, and allows the challenge of Western paradigms (Davis & Ruddle 2010). Brook & McLachlan (2005) argue that a balanced assessment of the strengths and weaknesses inherent in both biological data and LEK in isolation, and in contrast to each other, should be used to make decisions regarding the validity of the data. Including communities in this process can minimize the authority held by the researcher.

Although the inclusion of LEK and TEK in biological research is not without flaws where methods and best practices need to be streamlined (Davis & Wagner 2003, Brook and McLachlan 2008), many agree that using LEK and TEK in research is a valuable practice for both biologists and communities included in the process (Moller et al. 2004). Traditional knowledge involves the observation of events in the world without categorization. Thus, TEK includes awareness of information that may not have been incorporated into scientific studies in the area (Pierotti & Wildcat 2000). Effective science is often limited by time, monetary limitations and disciplinary requirements. Applying TEK or LEK can provide a means to overcome these limitations and provide greater insight into ecological phenomenon (Cook et al. 2013). There is a growing body of literature focused on promoting diversity in knowledges in research to improve ecological resistance by making management structures less rigid and more adaptive to change (Berkes et al. 2000b; Folke 2004; Berkes & Turner 2006). Although this concept can be problematic due to unequal power relations between researchers and knowledge holders (Nadasdy 1999), and is inconsistently applied due to varying understandings of the degree of knowledge integration required, the emphasis on novelty and innovation in resource management strategies is nevertheless a positive contribution to global conservation issues (Bohensky & Maru 2011).

2.3. Citizen science, participatory mapping and Geographic Information Systems

"Citizen participation is citizen power" - (Arnstein 1969)

Increasing the environmental and scientific knowledge base of individuals can help facilitate the development of a conservation ethic through increased ecological knowledge and place-based nature experiences (Berkes & Turner 2006; Dickinson et al. 2012). Citizen science is a branch of science that focuses on engaging non-professionals and non-experts in scientific research projects at all levels – hypothesis and research question derivation, data collection, and result interpretation, though the large majority of examples of citizen science have focused solely on data collection (Miller-Rushing et al. 2012). The goals of citizen science as a research strategy are to gain valuable information that may not be feasibly collected otherwise, and simultaneously encourage emotional investment in the natural world with education (Bonney et al. 2009; Jordan et al. 2012). The theory behind citizen science is based on the idea that participation can lead to citizen empowerment. In her classic work on citizen participation, Arnstein (1969) outlines 8 levels of citizen participation (Figure 2.1) which help move citizens toward further engagement and thus further power. Citizen science and its methods aim to move citizens to the top three levels. Policy grounded in public opinion that includes stakeholder perspectives can be instrumental is solving and deciphering the complex socio-economic environmental issues faced today by creating shared understanding and joint cooperative action (Brugnach et al. 2008; Garmendia & Stagl 2010).

As a research method, citizen science is highly effective at advancing scientific knowledge (Bonney et al. 2009) and is especially complementary to local hypothesis driven research (Dickinson et al. 2010). Typical citizen science projects could include documenting change in the historical and current range of a species, determining species abundance over a large geographic areas and confirming the presence of rare organisms (Lepczyk 2005; Dickinson et al. 2012). Citizen science helps scientists collect data and connect with people in places and at scales that would not otherwise be possible, making citizen science a unique contributor to environmental research (Dickinson et al. 2010). Correct project design and data validation is critical in citizen science projects (Bonney et al. 2009). If the data collection methods are well designed and communicated, data collected by participants should be robust and large sample sizes provide opportunities for cross-validation (Cohn 2008).

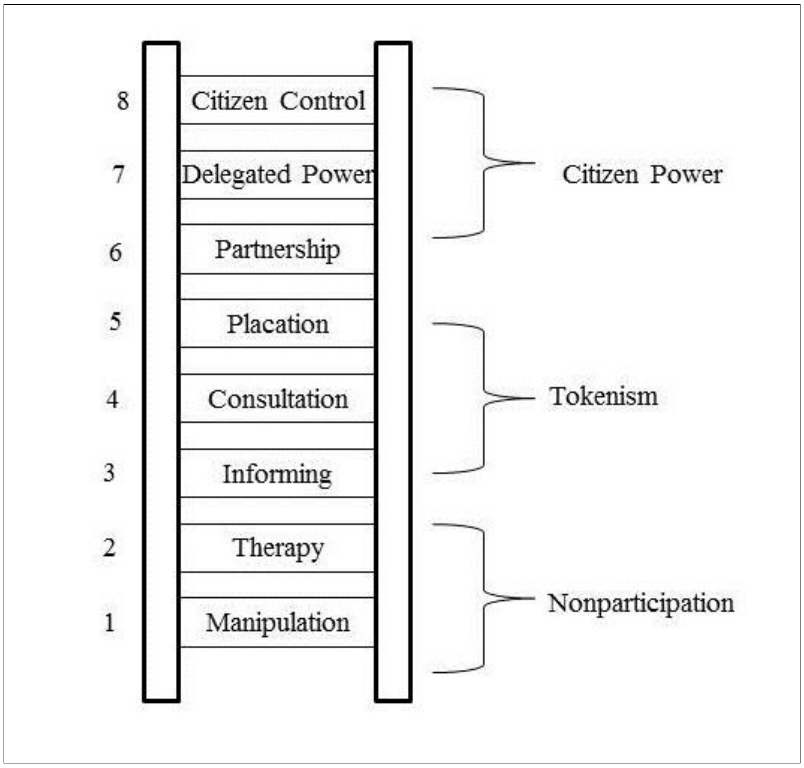

Figure 2.1 – The eight levels of citizen participation in governance and policy, as outlined by Arnstein (1969).

Participatory Geographic Information Systems (PGIS or participatory mapping), is one type of citizen science tool used by researchers. PGIS is used to facilitate the spatial translation of experiential knowledge held by non-scientists about an area or phenomenon to scientists (Tobias 2000). PGIS can be used to identify and document many spatial patterns, for instance, traditional land uses within an area, locations of hunted wildlife species or historical artefact caches (Clevenger & Wierzchowski 2001; Sieber & Wellen 2008; Smith 2008) . In PGIS, knowledge holders are asked to annotate a map which is either a physical or digital representation of the area, with locations of events, occurrences or phenomenon that help bring additional insight to a decision-making or research process (Jankowski 2009). PGIS has been used by researchers for approximately 20 years (McCall & Dunn 2012). In order to document the information gathered in participatory mapping sessions, a software program or Geographic

Information System (GIS) is used to input the data into a digital format, where it can analyzed and combined with other spatial information.

Participatory GIS is an approach to data collection that emphasizes community-led and context-driven research (Dunn 2007; Jankowski 2009). The PGIS process is often marketed as a mechanism to facilitate the involvement of local residents in the decision making process concerning issues that impact their community (Lewis 1995; Sandström et al. 2003; Jankowski 2009). Communities or groups of individuals give input and feedback into the decision making process, with the end goal of PGIS facilitating an outcome that is acceptable to both researchers and decision makers, and the impacted people. PGIS is advocated as a technique to challenge top-down methodologies that can ignore local knowledge and community participation (King 2002), and thus emphasize and reflect the multiple geographic realities that exist for different groups of people, in contrast to an objective and technical way of looking at the world (Dunn 2007).

Whether PGIS is truly participatory or not, and what it accomplishes, are key criticisms of the method. Participatory research can be defined as research where the local people in the study area take part in all the action components of the research process, in the planning, implementation and reflection process, which is the analysis and application of the results. Participatory research is a shift away from conventional research, where the power is held by the researcher who performs research on the subjects (Cornwall & Jewkes 1995). While PGIS projects are in name participatory, projects are often not truly participatory in the way defined above, as methods are often still dictated by the researchers, and often focus on community member engagement instead of creating a participatory process (King 2002). Although a chief tenet of PGIS and an often-reported result of the process is the engagement and subsequent empowerment of local knowledge holders, it is critical to note that an information holder is not necessarily a power holder. Indeed, some argue that the actual methods of PGIS do not consistently contribute to equity for the disempowered, (McCall & Minang 2005; Rambaldi et al. 2006) since outside researchers are still in charge of analyzing and presenting the data (Simpson 2001). Hence, PGIS is perhaps best thought of along Arstein’s ladder, beginning at the ‘consultation’ phase, with the potential to move up to ‘citizen control’.

An additional criticism of PGIS concerns that nature of the data itself. Abbot et al., (1998) state that using PGIS can create a sense of false accuracy and legitimacy as the data can be skewed by human perception of geography, and thus at times may not actually be correct or useful. They believe using PGIS data can fall into the trap of "Garbage in, Garbage out"(McCall & Minang 2005). However, others argue that this criticism warrants the question, "What degree of accuracy/precision are needed in PGIS?" and suggest that the error created by human perception may not influence results if an appropriate and systematic data collection method is followed (McCall & Minang 2005; Aditya 2010). Criticisms of both methodological and ethical issues can be mitigated by following structured protocols designed to encourage community participation and equitable knowledge sharing, as well as useful and accurate spatial information (Tobias 2000; Sieber & Wellen 2008). These protocols emphasize an ethical approach to data collection with clear guidelines and boundaries as to ownership of data and permission to use data, as well as a researcher mindset that is aware of the power dynamics intrinsic in group interactions and discussions (Rambaldi et al. 2006; Chambers 2006).

When studying wildlife, it is necessary to understand how and why populations are structured across the landscape, since the interactions between animals and the landscape they occupy are key determinants of an animal's fitness (Morris 2003). As such, the use of GIS-based spatial modelling techniques to understand these relationships is becoming a critical part of wildlife research (Alldredge et al. 1998; Gergel & Turner 2002). Using GIS and spatial modelling can document the interactions between a species, its predators, and environmental and human impacts (Wobeser 2002). For example, when monitoring disease outbreaks, spatial information can identify disease hotspot and the distribution of risk factors like artificial feeding stations or pinch points on the landscape (Pfeiffer & Hugh-Jones 2002; Wobeser 2002). When managing wildlife populations, GIS can be used to quantify habitat selection patterns, identify priority conservation areas, or document social interactions between individuals (Johnson et al. 2004; Balram et al. 2004; Vander Wal et al. 2012). GIS can also be used to help scientists and members of the public communicate information back and forth about wildlife populations (Huntington 2000; Michael 2003). PGIS in particular, can be useful for wildlife researchers when researchers have data that is limited in scope and in time, and can be a useful complement to traditional biological research methods which have their own biases and limitations (Lewis 1995; Johnson & Gillingham 2004; Brook & McLachlan 2009).

2.4. Mapping the spatial distribution of species

The spatial or geographical distribution of a species across a landscape is a critical component of ecology. Identifying species distribution and assessing the factors that determine distribution can determine ecological and evolutionary insights, and is a fundamental component of effective wildlife and resource management (Elith & Leathwick 2009; Higgins et al. 2012). Species distribution model or SDMs, are used to estimate and define distribution and have numerous applications (Higgins et al. 2012). Some examples of SDM applications are: understanding patterns of plant or animal occurrence across the landscape, thus guiding conservation efforts (Johnson et al. 2004; Nielsen et al. 2005; Johnson & Gillingham 2008); documenting species abundance across an area (Nielsen et al. 2005); identifying appropriate areas for reserve selection (Carroll et al. 2003); and to predict the impacts of human growth on biodiversity (Guisan & Thuiller 2005). SDMs work by combining species occurrence or abundance observations with environmental characteristics to quantify a relationship between these factors, and apply the relationship across the landscape (Elith & Leathwick 2009). Models commonly used by ecologists are static, where the models are created with the assumption that distribution is constant throughout the studied period. Modelling uses current statistical approaches to relate the distribution of species to its environment. When they are well-designed and properly implemented, SDMs reflect the natural distribution of a plant or animal species (Elith & Leathwick 2009) and can be used to predict future changes.

Many statistical techniques and modelling approaches can be used to create SDMs (Guisan & Zimmermann 2000) but most models use locations and measurements of animal presence or abundance, and sometimes their absence. These can include survey observations, historical museum records, or most commonly, radio-collar data (Loiselle et al. 2003; Frair et al. 2004). Presence and abundance datasets indicate where a species is found, but may not include where a species is absent, which presents a challenge for modelling. Pearce & Boyce (2006) outline several methods for estimating species distribution using presence-only datasets: (1) Use presence data points to roughly outline species distribution, (2) compare the presence points with pseudo-absence locations where species data is missing, (3) contrast presence locations with available (ie. random) locations, and (4) use data on relative species abundance to model abundance. When contrasting presence points with available points (method 3), several statistical options are available. Researchers can use an Ecological Niche Factor Analysis (ENFA) (Hirzel et al. 2002), case controlled logistic regression (Manly et al. 2002), logistic regression with an exponential model (also known as a resource selection function or RSF) (Manly et al. 2002), or a logistic regression algorithm, which is used to estimate a logistic discrimination model (Keating & Cherry 2004). Of the above techniques, those that use either pseudo-absences or available locations are generally thought to be the most effective (Barbet-Massin et al. 2012). RSFs are frequently used to model distribution when using locations are categorized as available (Smulders et al. 2010). However, as no model is perfect, when selecting a model for use, it is critical to consider the trade-off between model accuracy and generalizability, which differs between each technique (Guisan & Zimmermann 2000).

All SDMs rely on two standard assumptions (Guisan & Thuiller 2005): that the organism modelled is in pseudo-equilibrium with its environment (Guisan & Theurillat 2000), and that species occupy a specific niche unique to them, where the biological characteristics present in this niche can be used to determine distribution (Hutchinson 1957; Guisan & Zimmermann 2000). An organism is in pseudo-equilibrium with its environment if the organism lives in locations that are thought to be suitable and does not live in locations that are unsuitable (Araújo & Pearson 2005). However, in reality, species occurrence is rarely so simplistic, where non-equilibrium can occur in many biological scenarios, such as in a slow dispersing species or an invasive species (Václavík & Meentemeyer 2012). Hutchinson (1957) defined the fundamental niche as an "n-dimensional hypervolume", where every point within that hypervolume defines the environmental conditions where the species of interest could exist. The species should be able to occupy that area indefinitely, as long as the environmental conditions remain suitable. He also discusses the realized niche, which is smaller than the fundamental niche, where the niche a species occupies is limited by interspecies competition. A species only occupies niche space where it is the dominant competitor. Much further debate exists regarding which is the appropriate niche to model in SDMs and which niche definition is actually modelled using conventional methods (Araújo & Guisan 2006). Pulliam (2000) summarizes four common niche types that are represented in distribution modelling: (1) the fundamental niche, where a species occurs anywhere with the appropriate environmental conditions; (2) the realized niche, smaller than the fundamental niche and dictated by competition in addition to environmental attributes; (3) a source-sink relationship where a species continues to occur in a habitat type that does not adequately meet its needs, thus resulting in a population decline; and (4) dispersal limitation, where a species is not present in certain appropriate habitats because of reoccurring extinction events, but recolonization is limited by inadequate dispersal capabilities. In order to assess the accuracy and precision of SDMs, it is necessary to identify how close the modelled species are to equilibrium with their environment and determine which niche is being modelled, whether the niche is appropriate to the goals of the researcher and if the biological interactions that should be included are reflected in the model (Guisan & Thuiller 2005; Higgins et al. 2012; Wisz et al. 2013).

While SDMs should predict the distribution of a species accurately when data collection is precisely designed and relevant environmental predictors are used (Araújo & Guisan 2006), meeting these two conditions is difficult, as species presence data is often already collected or collected with a different purpose in mind (Soberon & Peterson 2005; Pearce & Boyce 2006). As well, identifying which environmental variables, both abiotic and biotic, that directly drive distribution is difficult because of the complexity of interactions that influence biological organisms (Austin 2007). Final developed models need to be accurate and reliable enough that they can be used by a variety of users, therefore, evaluation is necessary (Loiselle et al. 2003). Evaluation of SDM accuracy should provide the creator with information about where to improve model fit (Johnson et al. 2004) and also indicate some measure of generalizability of the model to other areas or other uses (Vaughan & Ormerod 2005). Another area of improvement in the development of SDMs is the inclusion of ecological theory and biological phenomenon and relationships in the model (Austin 2002, 2007; Higgins et al. 2012). Nevertheless, SDMs are a valuable and widely used component of species monitoring and management.

2.5. Resource selection functions

A fundamental assumption of ecology is that an organism will disproportionately select resources relative to resource availability, if that resource affects the organism’s fitness (Morris 2003). Differential resource selection is one the mechanisms that permits species to coexist simultaneously (Rosenzweig 1981). By quantifying resource selection, it becomes possible to model and extrapolate selection patterns by individuals, populations or species across the landscape (Manly et al. 2002). This information can then be used to understand and characterize long term resource requirements of the study species (Long et al. 2009), providing insight into such phenomenon as distribution, population abundance and competitive interactions.

Resource selection functions (RSFs) are a commonly used statistical model that make use of this assumption (Johnson & Seip 2008). In the simplest sense, an RSF depicts how organisms choose to meet their basic requirements for reproduction and survival, by statistically comparing the environmental characteristics in locations where an organism is found to the characteristics in locations where they are not found. It is then possible to quantify resource selection patterns and identify the probability of use of an area by the study organism (Manly et al. 2002; Boyce 2006).

Accurate creation, analysis and application of an RSF depends on the correct interpretation of the language used to describe the RSF (Hall et al. 1997). By defining what is meant by each term, other researchers and readers can ensure they are correctly applying the results of the research. In this context, habitat is the resources and conditions present in an area of occupancy of a given organism (Hall et al. 1997), while a resource is defined as a biotic or abiotic factor that impacts an organism's ability to live and reproduce (Thomas & Taylor 2006). Given that definition, a resource is anything that influences the ecology of an organism; some examples of possible resources include a food item, a topographical feature, a vegetation type or the presence of a predatory species. The term 'selection' refers to the process in which an organism chooses a resource, in contrast with the term 'preference', which is the choice of a resource independent of that resource's availability (Johnson 1980).

The statistical basis of the RSF is attractive because analyses and results are replicable, can systematically be validated and are spatially explicit (Boyce et al. 2002; Johnson et al. 2004). RSFs are generated by comparing the attributes of locations where an organism was present (used), with either the attributes of locations where an organism was not present (unused; i.e. a Resource Selection Probability Function; RSPF), or locations where an organism was assumed to have been absent but was not observed (available; Manly et al. 2002; Thomas & Taylor 2006). Most commonly, RSFs are created by using a form of logistic regression to compare datasets (McLoughlin et al. 2010), where the dependent variable is the used locations versus unused or available, and the independent variables are the resources (Boyce & McDonald 1999). In wildlife studies, the use-availability design is most commonly used (Johnson et al. 2006), as it facilitates the use of radio-collar data (Gillies et al. 2006; Aarts et al. 2008).

The RSF equation is as follows (Manly et al. 2002):

$$w(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \ldots \beta_k x_k)$$

where $w(x)$ is the relative probability of resource use at that unit, β_1 is the selection coefficient for the covariate 1 and x_1 is the resource value at that unit for covariate 1. The relative probability of resource use of the unit can be extrapolated for all units in the study area by inputting the values of the resource in the unit into the equation and using the derived relationship to estimate $w(x)$.

How RSF models are created, that is, which datasets are compared and how, is a subject often debated (McLoughlin et al. 2010). Keating & Cherry (2004) argue that with a use-availability design, logistic regression is an inappropriate method due to incorrect interpretation of the model results, partly because logistic regression assumes independence among observations which is not fulfilled when some available locations are actually used (Gillies et al. 2006). In these cases, the RSF values generated by logistic regression are not proportional to the probability of resource use by an organism, and thus researchers are more likely to commit a type I error (Gillies et al. 2006). Johnson et al. (2006) acknowledge that these points are technically correct, however, demonstrate that logistic regression derived RSF likelihood values are accurate enough to continue to use this design. Type I errors can also be avoided by using individual organisms as the unit of study, not total pooled relocations of all study organisms (Alldredge et al. 1998).

Correctly identifying the desired scale of analysis has large implications for both the creation and interpretation of an RSF (Gaillard et al. 2010). Organisms respond to environmental heterogeneity at different spatial scales, and ecological processes such as selection occur simultaneously at different ones (Wiens 1989; Boyce 2006). Johnson (1980) outlines four scales at which organisms hierarchically select resources. The first order is the range of an animal or the geographical boundaries of a species, the second order is the home range of an individual or group and third order is the use of resources within the home range, where home range is defined as the behaviours individual animals perform to survive and reproduce (Burt 1943). Expressed spatially, home range reflects the daily movement patterns of an animal ((Anderson et al. 2005a). Finally, fourth order selection is the food items an animal eats.

Once the scale of an RSF is chosen, it is important to determine both the grain (resolution) and extent (study area; Vaughan & Ormerod 2003) that will meet the research objectives, and ensure that the resource variables chosen reflect the correct scale, grain and extent (Meyer & Thuiller 2006). If differing scales of analysis are modelled, patterns of resource selection that are dictated by many scales may be inadvertently included in the analysis, leading to incorrect analyses and interpretations of results. Issues of correct selection of scale of study can be mediated by using a multi-scale analysis, combining scales in a scale-integrated RSF, conducting preliminary analysis to ensure the desired scale is the one being studied, or reporting results of analyses at multiple scales (Johnson et al. 2002; Boyce 2006; DeCesare et al. 2012). RSFs are a broadly applicable modelling technique that can be used to provide insight into scale-specific response, as well as other ecological questions (McLoughlin et al. 2010).

2.6. Landscape connectivity, corridors and least cost paths

How a species navigates through a landscape is dependent on landscape structure and species ecology. Assessing landscape connectivity and the ability of a species, its populations and sub-populations to move through and subsequently survive in the landscape, is a critical component of managing wildlife, especially in a heterogeneous, anthropogenic dominated landscape. Connectivity is defined as the "degree to which a landscape facilitates or impedes movement of organisms among resource patches" (Taylor et al. 1993; Tischendorf & Fahrig 2000). Conservation scientists generally agree that landscape connectivity promotes species and population viability (Gilpin & Soulé 1986; Beier & Noss 1998). However, the role that habitat corridors play in facilitating landscape connectivity is still debated (Beier & Noss 1998).

The concept of habitat corridors formally originated in wildlife research in the 1970s (Hobbs 1992) . Corridors have been defined many ways but can generally be thought of as native vegetation strips linking relatively large and intact patches of habitat surrounded by dissimilar (often non-native) vegetation that allow wildlife movement with the goal of increased population flow and viability (Hobbs 1992; Tischendorf & Fahrig 2000). Corridors and connectivity can be defined structurally, which involves how the landscape components are physically linked to each other, as well as functionally (Tischendorf & Fahrig 2000). Functional connectivity originates from the theory of island biogeography (MacArthur & Wilson 1967) and metapopulations (see Levins 1969; Hanski 1998). Functional connectivity is a species dependent measure of how landscape characteristics determine movement among resource patches" (Bélisle 2005). Corridors subtypes can also be more specifically defined based on structure, as well as movement patterns associated with the corridor (Hess & Fischer 2001).

Many researchers believe habitat corridors are a fundamentally positive contribution to wildlife population management (Hess & Fischer 2001) that promote biodiversity and mitigate the effects of fragmentation. Others argue that the fundamental assumption of corridors, that they encourage wildlife movement between isolated habitat types, which in turn reduces the extinction probability of the species, is hard to answer with certainty using the data available (Hobbs 1992). Harrison & Bruna (1999) found that corridors were only effective at helping large mammals move through the landscape but did not reduce the habitat degradation components of fragmentation, such as edge effects, while other studies determined corridors increase the movement of multiple species and facilitate pollination and seed dispersal (Tewksbury et al. 2002; Haddad et al. 2003). A review by Beier & Noss (1998) on whether corridors unequivocally promote connectivity found that no general answer fits all species and all landscapes. Rather, the success of corridors is highly context dependent. Additionally, habitat and thus corridor use patterns are species specific, making it difficult to define whether creating or maintaining corridors actually mitigates the effect of landscape fragmentation for the ecosystem overall (Fischer & Lindenmayer 2007). When identifying corridors, it is necessary to understand the ecology of the species to model functional connectivity appropriately (Chetkiewicz et al. 2006, Dugal 2013).

Various tools can be used to assess connectivity between remnant habitats and identify corridors. To estimate connectivity between habitat patches, the cost associated with movement between patches is calculated. This cost is typically a function of the distance between patches (Moilanen & Hanski 2001; Adriaensen et al. 2003) but more complex estimation techniques can also be used. Least cost path (LCP) modeling is a commonly used method that uses an algorithm to assign numerical cost values to a grid established across the study area (Gonçalves 2010). The cost assigned to each cell is determined by a set of attributes, where the combination of attributes that facilitate animal movement or provide favourable habitat have a low value, and the attributes that hinder or prevent movement are given a high value. The chosen attributes could be based on land cover, topography, or predation risk—essentially anything that inhibits the movement of the study species across the landscape. These factors and the resulting grid of assigned values is called a cost surface. The path of lowest cumulative resistance, or least cost path, is calculated through the study area from patches of interests and represents the path that costs the organism the least to travel across. This path can be used to distinguish corridors that wildlife are most likely to use to travel across the landscape. The accuracy of the predicted least cost path is, in part, based on the accuracy of the cost surface used (Clevenger et al. 2002). Expert opinion is commonly used to create cost surfaces, although recently more quantitative modelling approaches use models such as RSFs that explicitly link organism behaviour with landscape processes to create the cost surface (Chetkiewicz & Boyce 2009; Zeller et al. 2012, Dugal 2013).

While least cost paths are a valuable tool in conservation planning, current methods have several potential weaknesses. Criticisms of least cost paths tend to be based on lack of validation. Both lack of validation of the variables used to create the cost layer, lack of validation of model produced and path parameters such as width or length parameters that are arbitrarily defined and not justified, all also reduced the accuracy of the corridor generated in the least cost path (Pullinger & Johnson 2010; Sawyer et al. 2011). In addition, procedures and assumptions are often not the same between models, species and applications, where the uncertainty associated with models prevents comparisons between model conclusions and corridors identified. This uncertainty can reduce stakeholder buy-in to model results (Beier et al. 2008). A basic assumption in modelling pathways using a cost surface is that animals make movement decisions based on the same selection processes as habitat selection, even though there is currently little evidence that this is the case. To generate the best model possible, accurate and thoughtful research and decisions are required in each step when creating a resistance layer. Which environmental variables will be included and why, which biological data will be used to model the organism’s preference, and what technique will be used to convert this data into a cost surface are all decisions that need to be made (Zeller et al. 2012). While least cost paths may have some limitations, advantages like: flexibility of input data, and variety of potential applications such as dispersal scenarios and modelling the impacts of landscape change, make it a valuable and widely used tool for landscape conservation practices (Adriaensen et al. 2003; Sawyer et al. 2011).

2.7. Wildlife disease ecology and management

In the past two decades, interest in the study and management of wildlife disease has increased (Wobeser 2007). A potential causative factor of this trend is the risk of zoonotic transfer of wildlife disease into humans or to livestock, which has become a worldwide problem for human health and biodiversity (Daszak et al. 2000; Deem et al. 2001; Wobeser 2007). While disease in wildlife populations is a normal phenomenon, for populations that are already under stress or threatened, disease outbreaks can cause extinction or localized extirpation by killing hosts more rapidly than they can reproduce, or by reducing population growth rates, leaving populations more vulnerable to stochastic events (Lyles & Dobson 1993; Woodroffe 1999; Daszak et al. 2000; Wobeser 2002; Altizer et al. 2003). Importantly, disease in wildlife also frequently results in attempts to purposefully cull wildlife (Brook et al. 2014). By using a multidisciplinary approach which integrates individual, population and environmental perspectives of disease agent biology, the threat of disease can be somewhat mitigated (Daszak et al. 2000).

Effective disease management requires additional information than is provided by an epidemiological assessment of the infectious agent (McClintock et al. 2010). Researchers and decision makers need to consider the dynamics between agents, wildlife hosts and the environment variables, as all factors are interrelated and determine how successful a management program will be (Wobeser 2005). Lyles & Dobson (1993) articulate how a disciplinary separation between ecologists and veterinarians is resulting in ineffective disease treatment. For example, understanding host distribution, response to environmental change and sociality, as well as climate patterns and landscape topography are all crucial factors that influence disease transmission and prevalence (Deem et al. 2001; Pfeiffer & Hugh-Jones 2002; McClintock et al. 2010; Vander Wal et al. 2012) . Barlow (1996) advocates for an approach that includes some host ecology measures like host density-dependence in epidemiological models, while other argue for a totally interdisciplinary or transdisciplinary approach (Woodroffe 1999; McDonald et al. 2008; Brook & McLachlan 2009; McClintock et al. 2010).

Wildlife disease management is a complex process. Management decisions take place in an complex, unpredictable system where the results of an action are rarely guaranteed, and can in fact cause further problems (McDonald et al. 2008; McClintock et al. 2010). Managing disease in wildlife greatly differs from conventional domestic animal veterinary practice, as many domestic disease eradication strategies are not feasible or appropriate for intermixing and free ranging populations (Deem et al. 2001; Wobeser 2002). An example that illustrates the biological complexity of disease management in wildlife involves badgers *(Mele mele*) in the UK, the principal wildlife host of bovine tuberculosis (McDonald et al. 2008). Following culls to reduce disease prevalence for the benefit of livestock, badger ranging patterns changed. The change in badger behaviour compensated for any reduced transmission of bovine tuberculosis gained from population reductions (Donnelly et al. 2007).

Wildlife disease and human interactions with the environment are closely linked. On a global scale, many researchers believe that anthropogenic changes such as habitat loss, fragmentation, and climate change compound the negative impact of disease on wildlife populations, by either increasing disease prevalence or reducing population size (Woodroffe 1999; Daszak et al. 2000; Deem et al. 2001). Additionally, even seemingly small-scale actions performed by wildlife managers, such as animal translocations, can increase disease prevalence in an area (Cunningham 1996). However, disease in wild populations can also negatively impact people, directly through zoonoses such as rabies and by infecting livestock causing both indirect transmission of disease to human and associated socio-economic problems for agricultural producers and consumers (Daszak et al. 2000; Horan & Wolf 2005; Brook 2009). Human dimensions can also greatly influence the content and success of a disease management initiative (Riley et al. 2002; Decker et al. 2006). For instance, Peterson et al. (2006) found that long-term residents of a county in Idaho, USA were more likely to support lethal management options than newer residents, and were more likely to support state livestock agency actions while newer residents were more likely to support wildlife scientists. Other studies have shown that as disease prevalence increases, hunters are more likely to stop hunting host species, thus reducing the management options available (Needham et al. 2004).

Cervids pose a unique challenge for wildlife disease managers. In addition, to being an important recreational resource for hunters and wildlife viewers, they also have a close phylogenetic and ecological relationship with livestock, causing them to be of greater risk for disease transmission to livestock (Wisdom & Thomas 1996; Conner et al. 2008; Gordon 2009). Of the 8 most common diseases that infect North American elk, bovine tuberculosis and chronic wasting disease pose the largest threat to the prairie-parkland elk population (Bollinger et al. 2004; Conner et al. 2008; Brook et al. 2013).

Bovine tuberculosis is a zoonotic pathogen caused by the bacterium *Mycobacterium bovis* which originated from domestic cattle and can be transmitted between cattle and cervids (Conner et al.2008). Generally aerosol exposure or direct transmission is the most common means of bovine tuberculosis infection in cervids (Francis 1958). Infected animals exhibit respiratory disease detectable in the lymph nodes, tonsils and lungs. Intra- and inter-specific transmission of bovine TB also likely occurs via shared feeds (Gooding & Brook 2014). Elk in the Riding Mountain National Park region in Manitoba are one of two wildlife reservoirs for the disease in Canada (Lees 2004; Shury & Bergeson 2011). For the Riding Mountain elk population, the spread and prevalence of bovine tuberculosis has the potential to indirectly impact elk numbers by causing managed population reduction or eradication efforts, as an attempt to eradicate the disease. When elk leave the park boundaries and come in contact with cattle, they may infect the cattle (Brook & McLachlan 2009; Brook et al. 2013). Once livestock is infected, the entire herd must be destroyed to control the disease as per regulations of the Canadian Food Inspection Agency. Thus, the presence of elk for some cattle ranchers may be seen as negative and ranchers may be less inclined to protect and coexist with elk populations (Brook 2008).

Chronic wasting disease (CWD) is a neurodegenerative form of a spongiform encephalopathy caused by misfolded proteins known as prions (Williams & Young 1982; Williams et al. 2002). CWD spread into Canada from the United States in the period of 1996 – 2000 via imported ranched wildlife (Williams et al. 2002; Kahn et al. 2004), and as of 2013 has been identified in wild and captive cervids in 15 US states and 2 Canadian provinces (Alberta and Saskatchewan; Figure 2.2; (Kahn et al. 2004; Bollinger et al. 2004; Saunders et al. 2012; Canadian Cooperative Wildlife Health Centre 2013b). CWD infects both wild and domestic cervids and has a 100% eventual mortality rate (Williams & Young 1982; Williams & Miller 2004; Argue et al. 2007). CWD can be spread directly from animal to animal, or indirectly from the environment to an animal (Miller et al. 2006). Once in the environment, the infectious prion can exist and reinfect an animal after two years, in contrast to *Mycobacterium bovis,* which can last up to 18 hours in direct sunlight (Soparkar 1917; Mathiason et al. 2009). All currently known naturally susceptible host species (elk, mule deer (*Odocoileus hemionus*), white tailed deer (*Odocoileus virginianus*), and moose (*Alces alces*) live in the prairie and parkland region of Canada with a considerable spatial overlap among species distribution (Raymond et al. 2000;

Kahn et al. 2004; Williams & Miller 2004). In areas endemic with the disease, disease prevalence in populations seems to be increasing while the range of the disease is spreading spatially (Saunders et al. 2012). Due to the ability of prions to remain infectious for long periods in the environment (Williams & Miller 2004), the overlapping range between mule deer, white tailed deer and elk in the prairie-parkland region, the lack of an existing prophylactic treatment, and the current inability to accurately diagnose living infected animals; CWD has the potential to cause direct impacts and drastically decrease cervid populations (Bollinger et al. 2004; Conner et al. 2008; Saunders et al. 2012).

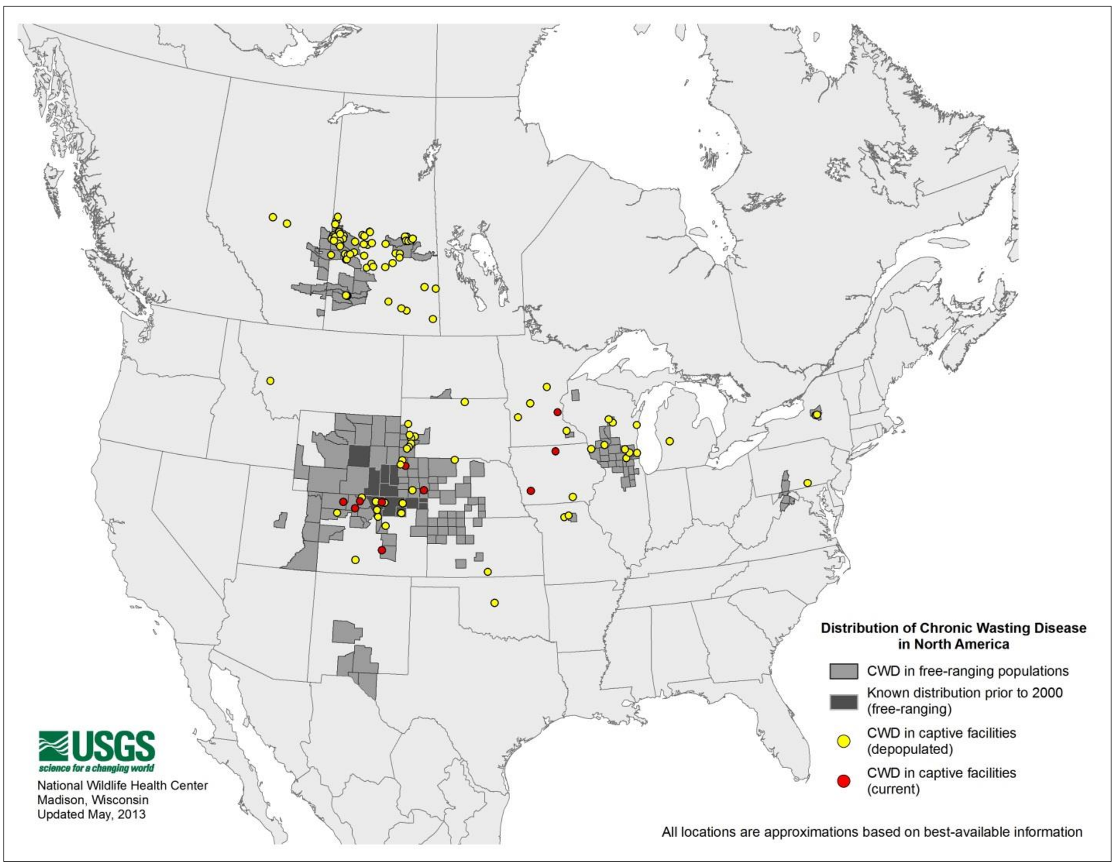

Figure 2.2 – Map depicting the current distribution of CWD in free ranging and captive cervid populations in North America (National Wildlife Health Center 2013).

CHAPTER 3: APPPLYING LOCAL ECOLOGICAL KNOWLEDGE WITH BIOLOGICAL RESEARCH TO MAP ELK DISTRIBUTION IN THE PRAIRIE-PARKLANDS OF CANADA

3.1 Abstract

While the use of local ecological knowledge (LEK) with biological research is frequently promoted as an opportunity for stakeholder engagement, LEK is rarely used quantitatively and is often seen as less accurate than conventional expert-based biological data. In this paper, I used elk locations derived from LEK, as well as radio-collar locations, to create a resource selection function (RSF) of elk distribution in the prairie-parklands of Saskatchewan and Manitoba in Western Canada. The objectives of this study were to develop an approach that effectively and respectfully used both LEK and radio-collar data, then apply this method to identify elk distribution. I held 10 participatory mapping workshops throughout the study area with 71 local experts who identified elk herd locations. The LEK data was digitized and used to identify sections ($1.6 \times 1.6\ km^2$) across the study area where elk were present. Radio-collar locations were also used to identify sections with elk (n = 328 collared elk; 1998–2012). Environmental predictor variables were used with elk presence sections to create RSFs which were extrapolated across the entire study area. Three RSFs were produced using the LEK-derived locations only, radio-collar locations only, and the combined locations from both datasets. In all three modelling approaches, elk showed strong avoidance for paved roads, selection for proximity to protected areas and selection for agricultural crops across all three models. When validated with an independent dataset of elk crop damage claims, the RSFs accurately predicted elk distribution in the study area. The three RSFs produced from the different approaches were not significantly different overall. This research demonstrated that LEK can be used to provide accurate and lower cost RSF models for wildlife over large areas and is an appropriate alternative method of data collection when estimating species distribution.

3.2 Introduction

In order to create effective conservation strategy, the geographic distribution of the species of interest must be known. However, determining species distribution patterns is often difficult, requiring a large data set of animal presence or abundance, and the extrapolation of small-scale occurrence patterns to landscape and regional scales (Loiselle et al. 2003). Distribution models combine species occurrence observations with predictive environmental variables and quantify a relationship between these factors, allowing the extrapolation of this relationship across the landscape (Elith & Leathwick 2009). Datasets that cover an entire study area are very expensive and are often collected for another purpose (Pearce & Boyce 2006); thus have some limitations when used to define species distribution. As such, it can be difficult to create quantitative distribution models that accurately reflect the ecological dynamics and behaviour of the species (Austin 2007). The inclusion of local ecological knowledge (LEK) as an occurrence dataset for a wildlife species is an alternative approach that can be used to determine species distribution patterns (Anadón et al. 2009)

Local ecological knowledge can be defined as the individual insights derived from personal observations and experience that occur when living and working in the natural environment (Gilchrist et al. 2005; Brook & McLachlan 2008). LEK differs from traditional ecological knowledge (TEK) which is defined as "all types of knowledge about the environment derived from experiences and traditions of a particular group of people" (Usher 2000). The validity of LEK is based on the concept that experts exist and are not solely created by formal education (Evans 2008), and therefore local experts can provide information that is valuable in its own right when answering biological questions. LEK provides complementary and revelatory insights as the knowledge held differs in spatial and temporal scale then knowledge gained by traditional, scientific biological research (Moller et al. 2004; Gagnon & Berteaux 2009). Including LEK in biological research can promote stakeholder involvement in management, and can shift power from researchers to the local people whose lives will be impacted by research outcomes (Agrawal 1995; Pierotti & Wildcat 2000). The use of alternative knowledge systems within wildlife management and research also promotes the valuation of non-Western ontologies, which may provide unique and previously unknown insights into biological crises (Agrawal 1995; Berkes et al. 2000a). The inclusion of LEK in biological research has increased in recent years (Brook & McLachlan 2008); however, LEK is less commonly used in quantitative estimates of wildlife distribution (Anadón et al. 2009). In particular, LEK has been infrequently used to create resource selection functions (except see Brook & McLachlan 2009; Polfus et al. 2014), which is a tool commonly used to generate species distribution models (Smulders et al. 2010).

Elk populations (*Cervus canadensis manitobensis*) in the prairie-parkland region of Canada in south-central Saskatchewan and Manitoba are threatened by habitat conversion to agriculture, disease and human-wildlife conflict (Bollinger et al. 2004; Conner et al. 2008; Brook 2009; Gordon 2009). Prior to human settlement, elk herds roamed freely across the Canadian prairie and parkland, but unregulated hunting and agricultural expansion caused a drastic range collapse (Bryant & Maser 1982; Wisdom & Cook 2000). Population estimates for Saskatchewan and Manitoba are 15 000 and 7350 respectively, with elk herds largely clustered in the transition zone between the prairie and boreal forest ecosystems. However, definition of elk distribution or associated population estimates in the prairie-parkland region have not yet been completed (Arsenault 2008; Manitoba Conservation 2013). Current published research on elk in the prairie-parkland region is largely limited to Riding Mountain National Park in southwestern Manitoba ((Brook 2010; Dugal et al. 2013; van Beest et al. 2013; Vander Wal et al. 2013) and around Cypress Hills Interprovincial Park in southwestern Saskatchewan (Hegel et al. 2009). Both of these study areas include large protected areas and adjacent agriculture, yet no research has been done on elk living within the agriculture-dominated prairie. As such, a quantitative distribution model for elk that includes habitat selection patterns in a variety of habitat types is required to support informed management and conservation decisions.

The purpose of this study was to develop an approach that facilitates the incorporation of local ecological knowledge with radio-collar data in a quantitative manner then use this method to map the regional scale distribution of elk in the prairie-parkland region of Saskatchewan and Manitoba, Canada. Specific study objectives were to: (1) generate spatial models of elk distribution using three different elk location datasets: LEK only, radio-collar data only and LEK and radio-collar data combined; (2) determine participant perceptions of elk habitat; (3) validate the three distribution models from each dataset with an independent dataset; (4) compare the three models in order to understand the strengths and weaknesses of each dataset when generating regional scale distribution models; and (5) consider how this approach of mapping species distribution could be applied to similar regional scale efforts at conservation and management.

3.3 Methods

3.3.1 *Study Area*

This study took place in the prairie-parkland region of Saskatchewan and Manitoba within the Boreal Plains and Prairie ecozones, with a total area of 614 091 km^2 (Figure 3.1). Study area boundaries were delineated based on the historical distribution of the Manitoban elk subspecies (Soper 1946; Bryant & Maser 1982; Polziehn et al. 1998; Laliberte & Ripple 2004). Climate, vegetation and human development extent drastically vary across the study area (Wiken 1986). The southern half of the study area is dominated by extensive agricultural development, mainly cereal and oilseed crops, and beef cattle (Brook & McLachlan 2006; Rashford et al. 2011). Current crop production estimates in the region are 9.5 million tonnes of canola, 18.6 million tonnes of wheat and 8.8 million tonnes of barley and oats (Statistics Canada 2013). Remaining forest in the southern portion of the study area is primarily trembling aspen (*Populus tremuloides)* in small and largely fragmented patches. Small wetlands or potholes are also present throughout the region, though the majority have been lost to agricultural expansion (Ecological Stratification Working Group (Canada) 1996; Rashford et al. 2011). Over the last 150 years, 83% of native grassland in the region has been replaced by agriculture (Samson et al. 2004). Other ecosystem changes, such as a decline in numbers of large herbivores, and a drastically reduced fire frequency have further altered historical prairie grassland ecosystem dynamics (Samson et al. 2004; Rashford et al. 2011). Native grass types include blue gramma (*Bouteloua gracilisi)*, wheat (*Triticum* spp.) and spear grass (*Aristida* spp.), which are distributed across the rolling plain. Wildlife such as white-tailed deer (*Odocoileus virginianus*), coyote (*Canis latrans)*, moose *(Alces alces)*, wolves *(Canis lupus)* and elk continue to make use of the modified landscape (Stewart et al. 2002; Côté et al. 2004; Gooding & Brook 2011)

Along the northern edge of the study area, the boreal transition zone was historically forested with trembling aspen and balsam poplar (*Populus balsamifera)* with interspersed black spruce (*Picea mariana)*, jack pine (*Pinus banksiana)* and tamarack (*Larix laricina;* Ecological Stratification Working Group (Canada) 1996). Increased resource extraction by the oil, gas and forestry industries, in addition to agricultural expansion, has greatly changed much of the region and fragmented the remaining forest (Archibold & Wilson 1980; Ecological Stratification Working Group (Canada) 1996; Hobson et al. 2002).

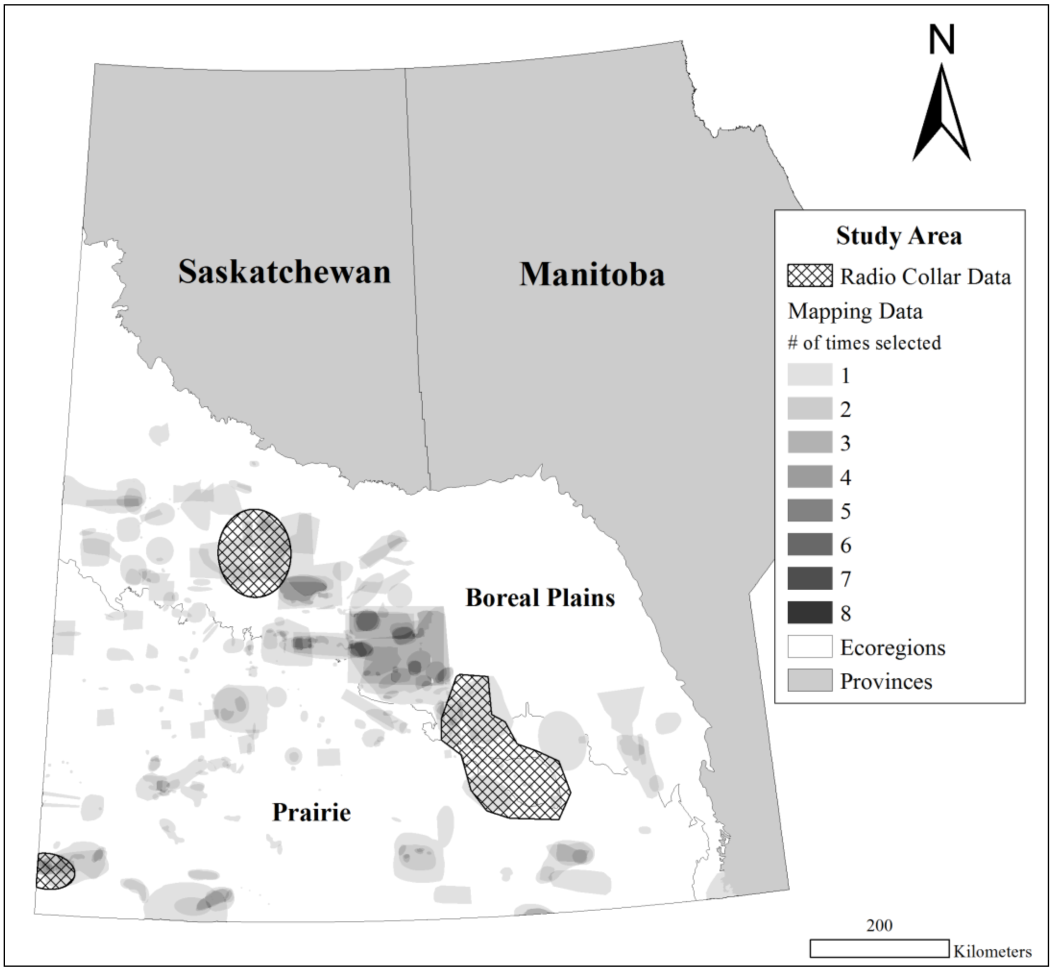

Figure 3.1 – Study area in central Saskatchewan and Manitoba, Canada and elk locations as identified by local ecological expert participatory mapping and radio-collar data. The mapping data is displayed by the number of times local experts identified that location as an elk herd presence point. The general range of each radio-collar location is depicted as hatched areas. The lower left dataset is in Cypress Hills (*n* = 64; 1998–2000), the central locations are from the Montreal Lake dataset (*n* = 18; 1999–2000) and the locations to the right belong to the Riding Mountain dataset (*n* = 246; 2002–2012).

3.3.2 *Participatory Mapping Workshops*

In order to obtain information on the distribution of elk in the study area, I held 10 local ecological knowledge workshops between June 2012 and February 2013, consisting of: two structured daylong workshops, two meetings with regional conservation officers, two individual sessions with provincial biologists and four informal sessions with licensed hunters. Four of these sessions were held with individuals from Manitoba and the remaining six were held with individuals from Saskatchewan. All sessions excluding the two workshops were between 30 minutes and two hours. I received ethics approval for this work from the University of Saskatchewan (BEH 12-151).

Knowledge-gathering session participants were all local experts, which I defined as individuals with experiential and/or academic knowledge of elk populations within the study area. Participants included Aboriginal people, conservation enforcement officers, provincial government biologists and licensed hunters. Prior to collecting data, participants were asked to self-evaluate their knowledge, and were given the choice to withdraw from the session if they did not feel they would make a significant contribution. I did not include results from individuals who withdrew. Participants were informed of their rights as a knowledge contributor, as well as how their knowledge would be used. A discussion regarding consent ensured participants understood what they were agreeing to and all participants gave written consent. Additionally, participants in the two full day workshops were compensated for mileage and attendance.

After a brief introduction to the project, its goals and an explanation of the process, a participatory mapping exercise was conducted at each workshop. The maps consisted of satellite land cover images of the study area at a 1:1 000 000 and 1: 2 250 000 scale covering the entire study area. The maps included major roads, towns and provincial and national parks so that participants could orient themselves. I asked participants to identify locations where elk herds were regularly located, and to mark these spots with as much precision as possible using a permanent marker. I encouraged participants to mark as many locations as possible and to discuss their answers with other participants. During the process, I also recorded stories and other comments from participants.

The second component of each workshop consisted of evaluating perceptions of how environmental attributes affect elk distribution and abundance in the study area. Participants in the first three workshops provided observations on what they perceived to be important habitat variables that determined elk distribution and abundance in the study area. These were compiled into a comprehensive list. The participants in the following five knowledge gathering sessions completed a survey on elk habitat attributes, which included the environmental attributes the previous participants had identified as important. The survey listed 10 habitat attributes such as "Easy access to forest cover" and "Highly varied topography (lowlands and hills)". Participants were asked to rank each attribute between 1 and 10 in order of absolute importance for elk, 1 being the most important and 10 being the least. They then indicated how each habitat attribute impacted elk populations, given the choices “positive”, “negative” or “undecided”.

3.3.3 *Radio-Collar Data*

I used elk locations from three existing radio-collar datasets distributed across the study area. Data collection used two different types of radio-collars: Very High Frequency (VHF), which uses a transmitted radio signal to demonstrate the position of the animal to an individual using a radio antenna to receive the signal, and Global Positioning System (GPS), which saves the animal locations to a data logger on the collar using positions determined by satellites. The datasets were located in Cypress Hills (SK, n = 64 VHF collared females; Hegel et al. 2009), Montreal Lake (SK, n = 18 VHF collared females), and the Riding Mountain Region (MB, n = 212 VHF collared; 120 females and 92 males) and 34 GPS collared (24 females and 10 males; Brook & McLachlan 2009; Vander Wal 2011; Brook et al. 2013). The Cypress Hills dataset was collected from 1998 to 2000, the Montreal Lake data from 1999 to 2000 and in the Riding Mountain Region (RMNP), I used locations collected between 2002-2012. Collar accuracy estimates were performed using stationary collars and quantified the variation between the location recorded and the actual, known location. GPS collars were accurate within 14 m and VHF collars were accurate to 18 m (Brook & McLachlan 2009).

To minimize errors in satellite collar locations, I excluded all locations taken < 2 days post-capture. I checked GPS locations for spatial errors such as impossible movements using the method outlined by Bjørneraas et al. (2010) which uses a combination of distance travelled and angle between locations to identify probable errors. A total of 0.1% of the dataset was removed based on these criteria. Animal capture was approved by the University of Manitoba Ethics board (#F01-037) and University of Calgary's Animal Care Committee (Protocol BI2001-065), and adhered to the guidelines in the Canadian Council on Animal Care (2003; Brook & McLachlan 2009; Hegel et al. 2009).

3.3.4 *Data Processing and Analysis*

A common method used to assess resource use by a population and identify distribution is a Resource Selection Function (RSF). An RSF estimates the relative strength of avoidance or selection of a resource or resources by comparing use and non-use of sites by animals with the equation: $w(x) = \exp(\beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \ldots \beta_k x_k)$, where $w(x)$ is the relative probability of resource use at that unit, β_1 is the selection coefficient for the covariate 1 and x_1 is the resource value at that unit for covariate 1. In this study, I followed a use-availability design, which is commonly used in wildlife studies with collar data, where absence locations are impossible to identify with certainty, leading to error asymmetry (Manly et al. 2002; Boyce et al. 2002; Johnson et al. 2006). I categorized radio-collar and participatory mapping locations as used sites. All elk locations identified in participatory mapping sessions were hand digitized into spatial polygons using a Geographic Information System (GIS; ESRI 2011), while radio-collar locations were plotted (Figure 3.1). Using the scale of the printed maps, and the thickness of the markers used to circle elk locations, I estimated the spatial error in the mapping process to be 288 m.

I scaled all input data for the RSF to individual sections. Sections are a unit used in land surveys throughout the Canadian Prairies, based on a measure 1.6 $\times$1.6 km^2 (1 mile by 1 mile). Identifying the scale of analysis has important implications for both the creation and interpretation of an RSF (Gaillard et al. 2010). Animals respond to environmental heterogeneity at different spatial scales, while ecological processes occur simultaneously at varying scales (Wiens 1989; Boyce 2006). Using sections as a spatial unit allowed me to assess elk selection on a broader landscape scale. The study area contained 89 960 sections. If a mapping location or radio-collar point was located in a section, that section was categorized as used. I also scaled the number of locations identified by section. For example, if multiple participants separately identified the same section as an elk location, that section was included in the analysis the corresponding number of times. In this way, sections that were identified by multiple participants were weighted more heavily than those identified once. An equal number of available sections (1:1 ratio) were randomly selected within the study area.

I hypothesized that 15 potential environmental predictor variables would influence the presence of elk at the regional scale in the study area (Table 3.1). These variables were derived using peer-reviewed literature and from the participant surveys of important environmental attributes. Elk distribution patterns are typically influenced by vegetation, land use, topography, and human influences on the landscape (Mao et al. 2005; Anderson et al. 2005b; Brook 2010; Baasch et al. 2010). As such, I used variables representing these key factors in the analysis. I obtained vegetation and land use for the study area from the 2000 Land Cover Vector Layer

(Natural Resources Canada 2009). To depict topographic variability, I used a 50x50m cell size digital elevation model (1:250,000; Natural Resources Canada 2000) and developed a Vector Ruggedness Measure of the landscape with a neighbourhood of 13 (Sappington et al. 2007). I calculated paved and unpaved road density using national road layers (Natural Resources Canada 2007). To generate human population density within the study area, I used population counts in census subdivisions from the 2011 Canadian census data (Statistics Canada 2011). To characterize protected areas in the study area, I used layers representing Manitoba provincial forests, Saskatchewan provincial forests and protected areas >1000 hectares (Government of Canada 2008; Manitoba Conservation 2010; Manitoba Conservation Forestry Branch 2010; Information Services Corporation of Saskatchewan 2012). To scale all variables appropriately, I used the percent of each land cover variable per section, and calculated the mean value per section for other variables. The values of all variables were scaled between 0 and 1. All predictor variables were screened for collinearity using a Spearman rank correlation matrix for all possible variable pairs with a cut-off of $r > 0.7$.When $r > 0.7$ in a variable pair, the more important variable was retained. Due to collinearity with crop, unpaved roads were removed from the analysis. I then screened remaining variables for multicolinearity using variance inflation factors and variable cluster analysis (Harrell Jr 2012).

3.3.5 *RSF Model Creation*

I developed three unique RSFs using three different datasets: mapping data only, collar data only and a combined dataset with all locations from each dataset. The combined dataset included all the elk location sections identified by both the radio-collar and local knowledge datasets. All statistical analysis used the R environment for statistical computing (R Development Core Team 2011). For all RSFs, I identified 10 *a priori* models hypothesized to predict elk distribution (Table 3.2). I assessed model fit using the difference in Akaike's information criterion adjusted for small sample size (ΔAICc) to identify the most parsimonious *a priori* model based on the model's ability to predict elk use, where the lowest ΔAICc indicates the model with the best fit (Burnham & Anderson 2001; Boyce et al. 2002). I then employed multi-model inference (R package MuMIn) to incorporate all models simultaneously and identify the cumulative influence of each predictor variable (Burnham & Anderson 2002; Barton 2013).

Table 3.1 – Description of environmental predictor variables used in three RSF models of elk distribution in the central Saskatchewan and Manitoba, Canada. Vegetation layer descriptions adapted from Wulder & Nelson (2003).

Environmental Covariate	**Description**
Coniferous Forest	% of section containing coniferous forest where 75% of the basal area is covered by coniferous trees such as jack pine or black spruce, 10–100% crown closure
Crop	% of section containing annual agricultural cereal, pulse, and oilseed crops
Deciduous Forest	% of section containing deciduous forest where 75% of the basal area is covered by deciduous trees such as trembling aspen and balsam poplar, 10–100% crown closure
Distance to Park	Mean distance per section to provincial and national parks and provincial forests
Forage	% of section containing perennial cropland and pasture
Grassland	% of section containing mixed native and tame grasses and herbs with <10% shrub cover
Herb	% of section with greater than 20% cover of vascular plants without a woody stem such as grasses or forbs
Human Population Density	Mean human population density per subdivision in each section
Mixed Wood Forest	% of section where no more than 75% of the area is classified as either coniferous or deciduous forest, 20–100% crown closure
Paved Road Density	Mean density of paved roads with a township per section
Shrub	% of section containing at least one third vegetation classified as a shrub such as willow *sp*. and dwarf birch (0-5m), crown closure 20–100%
Topographic Variability	Mean terrain ruggedness in each section
Unpaved Road Density	Mean density of unpaved roads with a township per section
Water	% of section containing water features
Wetland	% of section containing land cover classified as swamps, marshes, bogs, or fens where the water table is near, at, or above the soil surface, allowing time for wetland or aquatic processes to occur

Table 3.2 – *A priori* models hypothesized to predict elk distribution in central Saskatchewan and Manitoba, Canada and differences in ΔAICc for three different datasets of elk locations: a radio-collar dataset (*n* = 328 animals, 1998–2012), local ecological knowledge (LEK) participatory mapping locations and a combined dataset. The bolded model is the best model as indicated by a ΔAICc of 0.

		LEK	Collar Data	Combined Datasets
Model	*A priori* Models	ΔAICc	ΔAICc	ΔAICc
1	**Park+Crop+Forage+Grassland+Conifer+Mixed Wood+Deciduous+Herbs+Paved Roads**	**0.0**	**0.0**	**0.0**
2	Park+Crop+Forage+Conifer+Deciduous+Mixed Wood+Grassland+Wetland+Water+Paved Roads+Human Population	1188.1	26.2	24.6
3	Park+Crop+Forage+Grassland+Conifer+ Deciduous+Wetland+Shrub+Herb+Human Population	1719.0	183.2	303.1
4	Park+Crop+Conifer+Mixed Wood+Deciduous +Forage+Water+Human Population	2708.1	58.2	109.5
5	Park+Crop+Forage+Conifer+Mixed Wood+Shrub +Water+Grassland+Paved Roads+Topographic Variability	3023.2	106.2	166.1
6	Park+Crop+Forage+Conifer+Mixed Wood +Deciduous+Wetland+HumanPopulation+ Topographic Variablity	3660.0	17.0	24.6
7	Park+Conifer+Paved Roads+Grassland+Water+ Wetland	5720.5	78.8	166.9
8	Park+Conifer+Water+Grassland+Topographic Variablity+Human Population	7624.0	198.6	332.7
9	Park+Crop+Conifer+Mixed Wood+Human Population+Paved Roads+Topographic Variablity	8155.7	289.7	455.2
10	Park+Conifer+Grassland+Paved Roads	8973.1	409.2	580.8

I used a generalized linear model with binary logistic regression and the best *a priori* model to determine estimated beta coefficients values for each environmental predictor variable. The resulting beta coefficients were used to generate predictive RSF models. These models were mapped across the study area in ArcGis10 using sections as the unit of spatial resolution (ESRI 2011). All model values were rescaled between 0 and 1 by dividing the RSF value of each section with the maximum RSF value to allow comparison between the three models.

3.3.6 *Model Evaluation*

I assessed and compared the predictive ability of the three models produced with each datasets (LEK/collar/both combined). When evaluating models, I recognized that each dataset has intrinsic strengths and weaknesses that vary between datasets, for instance, the ability to detect hourly elk movements or opportunities for explicit community engagement (Brook & McLachlan 2005). The purpose of the comparison was to individually assess the models in the context of this project, and identify overall strengths and weaknesses of each. No one model was considered to be the 'truth' or used as a baseline to assess the value of another model (Brook & McLachlan 2005). I used a raster difference map to highlight areas where the radio-collar and mapping derived models differed in their prediction of elk distribution. To evaluate the predictive ability of the RSFs produced, I compared RSF scores to an independent dataset of elk crop damage locations. The dataset consists of 11 589 locations from 1993– 2012 across Saskatchewan and Manitoba where elk have significantly damaged harvested, standing or seeded crops (SCIC 2011; MASC 2012). The crop damage data covers areas where agriculture is present in Manitoba and Saskatchewan and includes a wide range of habitats. Using an independent dataset to validate the predictive ability of a model is the least biased way to assess the accuracy when modelling the landscape scale distribution of a species (Austin 2007; Elith & Leathwick 2009; Johnson & Gillingham 2008). Crop damage claims have been successfully used to differentiate habitat selection between cervid species (Sorensen 2014). Farmers can accurately identify which species is responsible for damage and the large size of this dataset minimizes the impact of any incorrectly identified locations (Brook 2009). To compare crop damage locations to RSF values, I created 10 evenly weighted bins of RSF values for each model. If the RSF has high predictive power, the number of crop damage claims should be largest in the higher ranked habitat bins and smallest in the lower ranked bins (Boyce et al.

2002). I used Spearman's rank correlation to assess this relationship. To compare the models produced against each other, I took a random sample of RSF values in 10% of the study area for all three models (n = 26 988 sections). The models were considered to be equivalent if the correlation between the datasets was significant ($p < 0.05$; Brook & McLachlan 2009).

3.4 Results

3.4.1 *Elk locations and local ecological knowledge*

I combined locations identified by radio-collars and LEK into one model by including all sections that were identified as used in each dataset. All local ecological knowledge gathering sessions resulted in 30 annotated maps, with a total of 392 locations identified and 98 556 used sections in the study area. The radio-collar dataset identified 1751 sections as used. Participants demonstrated a detailed knowledge of elk in the study area and were able to state with confidence specifically what elk in their area were doing. For example, one participant indicated that, "[The elk] go back and forth between the forest and agriculture. If it is really wet, they will go into the hills", and "If you take a map of the forest fringe across the province [of SK], that's where your elk are". The habitat variables identified by workshop participants and ranked by the survey takers were similar to the habitat variables that were included in the *a priori* models based on literature (Table 3.2).

3.4.2 *Elk distribution*

The RSF models using the different LEK and collar datasets identified elk distribution across the study area (Figure 3.2). For all RSF models, I identified *a priori* model one (ΔAICc) with 10 predictor variables (Table 3.2), as the model to best predict elk distribution. Across all three models, elk exhibited similar patterns of selection and avoidance for 9 out of 10 variables. For coniferous forest, the one variable that was not consistent, in the LEK based model, elk slightly avoided coniferous forests (β= -0.15, S.E = 0.03) while in both the radio-collar based (β= 1.35, S.E. = 0.26) and combined model (β= 0.48, S.E. = 0.16), elk selected for coniferous forests; (Figure 3.3). When looking at the impact (cumulative AIC weights) of each variable, I used a cut-off of $w_+ > 0.5$ to designate variable importance, where variables with w approaching 1.0 having a greater influence on presence of elk in the study area. For the LEK-derived dataset, 12 of the 14 variables were found to be important ($w_+ > 0.5$). In the radio-collar dataset, 9 variables were important and for the combined dataset, I found 13 of the 14 to be important. Participant surveys were consistent with the models about the impact (negative or positive for elk) of five habitat attributes on the probability of elk occurrence and differed about the impact of three habitat attributes (Table 3.3).

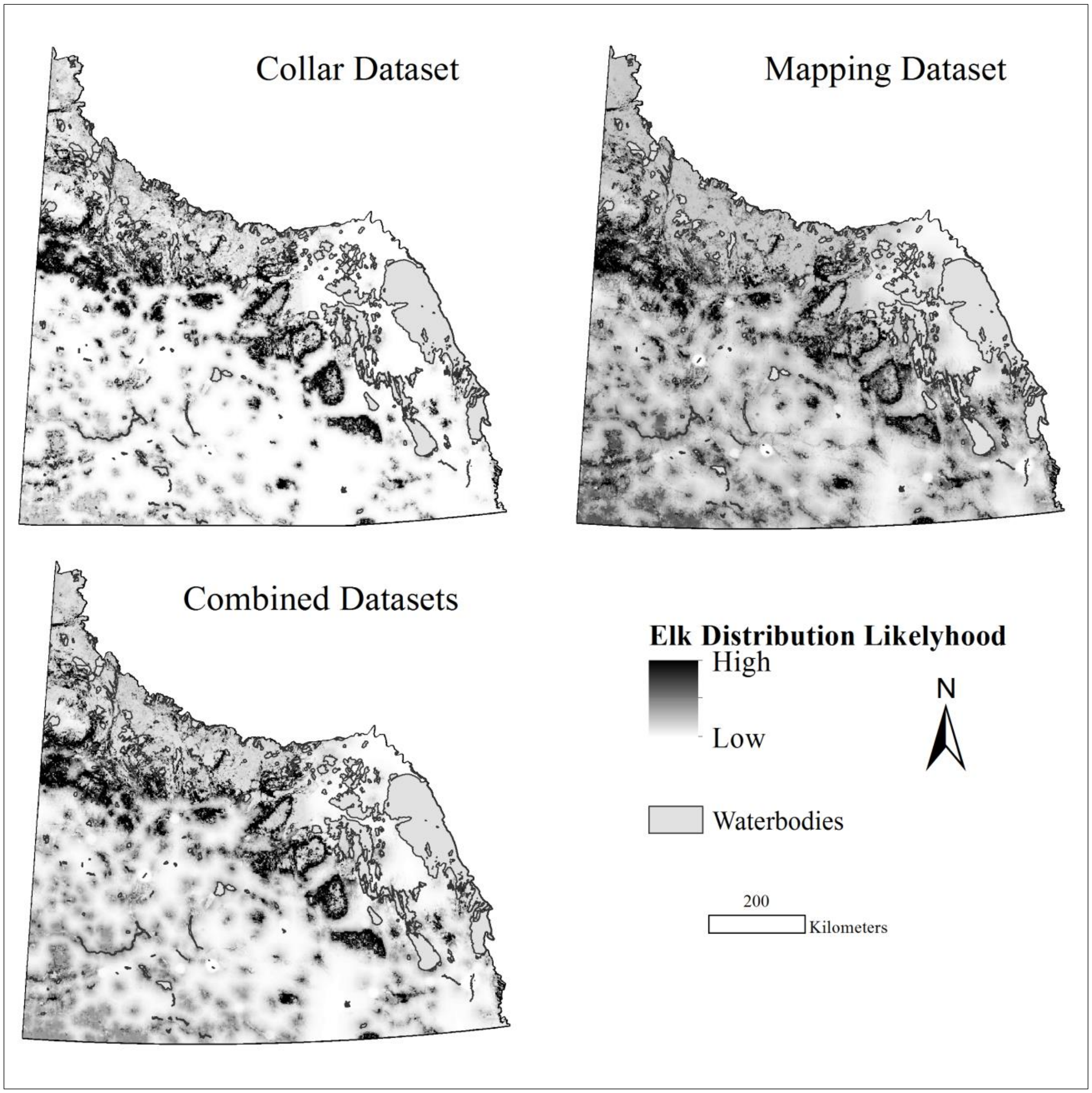

Figure 3.2 – Predicted relative probability of elk occurrence across southern Saskatchewan and Manitoba, Canada based on resource selection functions created with three datasets of elk locations: a radio-collar dataset (*n* = 328 animals, 1998–2012), local ecological knowledge (LEK) participatory mapping locations, and a combined dataset. Black represents higher probability of occurrence, while lighter colours represent lower probability.

Table 3.3 – Relative importance (w_+), β coefficient estimate with standard error, and survey results of environmental predictor variables of 10 environmental predictor variables hypothesized to determine elk distribution in central Saskatchewan and Manitoba using three different datasets of elk locations from a radio-collar dataset (*n* = 328 animals, 1998–2012), local ecological knowledge (LEK) participatory mapping locations and a combined dataset. Social survey results were gained from a survey given to 40 local knowledge gathering session participants.

Environmental Predictor Variable	Combined Datasets			LEK			Collar Dataset			Social Survey Results	
	w_+	β	S.E.	w_+	β	S.E.	w_+	β	S.E.	Importance of Habitat Type	Effect on Elk Presence
Coniferous Forest	**0.83**	-1.71	0.78	**1.00**	-0.80	0.12	**0.94**	-1.77	1.02	1	Positive
Deciduous Forest	**0.63**	1.30	0.91	**1.00**	1.34	0.12	0.47	0.90	1.39	1	Positive
Forage	**0.91**	0.28	1.00	**1.00**	0.96	0.12	0.42	1.18	1.38	7	Positive
Herb	**0.64**	1.58	1.02	**1.00**	1.70	0.13	0.47	1.78	1.62	-	-
Distance to Park	**1.00**	-6.27	0.33	**1.00**	-2.98	0.05	**1.00**	-16.27	1.04	2	Positive
Paved Road Density	**1.00**	-26.77	3.71	**1.00**	-14.74	0.56	**1.00**	-36.33	7.37	6	Negative
Shrub	**0.84**	-1.99	0.84	**1.00**	-0.99	0.13	**0.97**	-4.02	1.19	-	-
Topographic Variability	**1.00**	2.99	0.58	**1.00**	1.40	0.17	**1.00**	4.13	0.81	3	Undecided
Water	**0.99**	-2.11	0.95	**1.00**	-0.89	0.12	**0.94**	-2.87	0.87	5	Undecided
Mixed Wood Forest	**0.62**	1.26	0.93	**0.98**	0.43	0.10	0.47	1.25	1.33	1	Positive
Grassland	**0.91**	-0.71	0.99	**0.98**	0.46	0.11	**0.93**	-2.09	1.00	7	Positive
Wetland	**0.83**	-1.76	0.78	**0.95**	-0.35	0.10	**0.93**	-2.83	0.88	5	Undecided
Human Density	0.30	-2.65	7.36	0.35	-0.58	0.70	0.31	-10.74	20.48	-	-
Crop	**0.90**	-0.45	0.99	0.33	-0.01	0.20	**0.97**	-0.70	1.09	4	Positive

All variables with $w_+ > 0.5$ are bolded.

[a] Cumulative AIC_c weight of a variable, which is calculated by summing the AIC_c weights of all possible models containing that variable.

[b] Since surveys were participants derived, habitat attributes in the survey do not exactly correspond with environmental predictor variables. Some predictor variables were combined into one habitat attribute and some predictor variables were not included in the survey.

[c] Participant responses were pooled. The ranking of importance of each habitat type was determined by summing total responses. I assigned the effect of each habitat attribute on elk presence by using the option (positive, negative or undecided) that the majority of participants selected

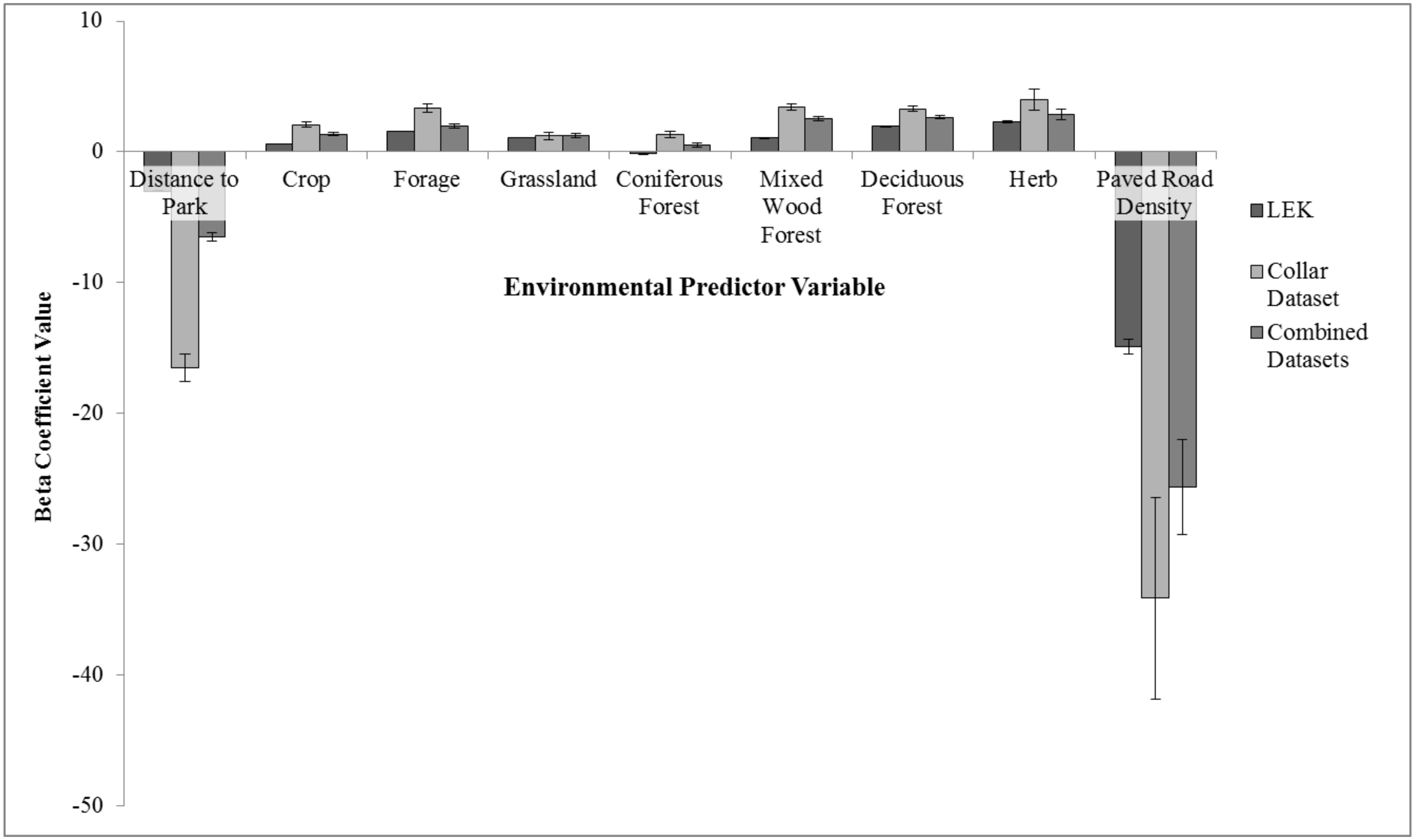

Figure 3.3 - Estimated β coefficient values with standard error for three RSF models of elk distribution in central Saskatchewan and Manitoba, Canada. The models were made using three datasets depicting elk locations: a radio-collar dataset (n = 328 animals, 1998–2012), local ecological knowledge (LEK) participatory mapping locations, and a combined dataset.

3.4.3 *RSF model evaluation*

All models had high predictive capacity, with rho's from the Spearman's rank correlation values of 0.97 between crop damage claims and the mapping dataset, of 0.98 between the radio-collar dataset and the crop damage claims and of 0.98 between crop damage claims and the combined dataset (Figure 3.4). Linear regressions between all model combinations indicated that models were not significantly different overall. The least similar models were the LEK and radio-collar datasets (r_s=0.69, d.f.= 26 988, $p < 0.01$), where the model produced with the combined dataset was more similar to both (LEK and combined dataset: r_s = 0.83,d.f. = 26 988, $p < 0.01$; radio-collar and combined dataset: r_s = 0.93,d.f. = 26 988, $p < 0.01$). The difference map that identified differences in RSF values from the collar and LEK-derived models (Figure 3.5) indicated that the models were largely consistent but differed in the southwest corner, agricultural region and along the southern portion of the forest fringe of the study area.

3.5 Discussion

This study demonstrated that LEK research can be used to create an accurate map of landscape scale elk distribution. This research also determined that RSFs produced using LEK only, radio-collar data only and a combined dataset were all accurate when validated using an independent dataset. Additionally, there was no significant difference between the distributions generated. The lack of significant difference between RSFs produced with LEK and radio-collar data indicates that LEK can be an effective alternative to conventional expert-based wildlife research techniques.

Thus far, only three peer-reviewed studies have created RSFs with traditional or local ecological knowledge. Brook & McLachlan (2009) integrated participatory mapping with radio-collar locations to identify cattle pastures with elk to create a predictive RSF of elk using cattle pastures. Polfus et al. (2014) created separate radio-collar based and traditional ecological knowledge derived RSFs and compared them. They found that both models accurately estimated caribou habitat.

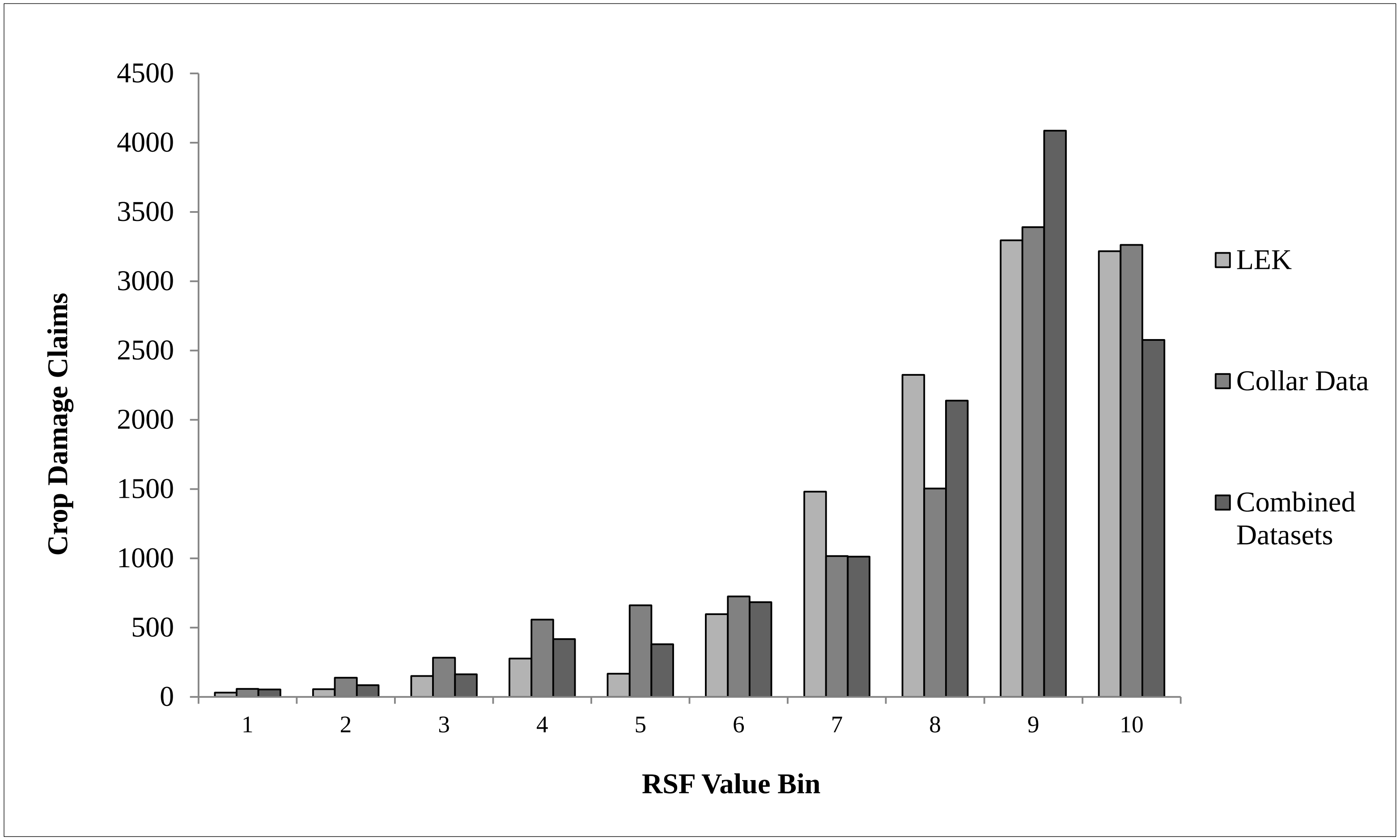

Figure 3.4 – Number of crop damage claims (*n* = 11 589; Saskatchewan and Manitoba; 1993–2012) in each RSF value bin derived for each RSF model produced. Three RSFs were generating depicting elk distribution in central Saskatchewan and Manitoba, Canada. Elk locations used in the RSFs were from a radio-collar dataset (*n* = 328 animals, 1998–2012), local ecological knowledge (LEK) participatory mapping locations, and a combined dataset. A higher bin number signifies a higher RSF value and thus a larger probability of elk occurrence.

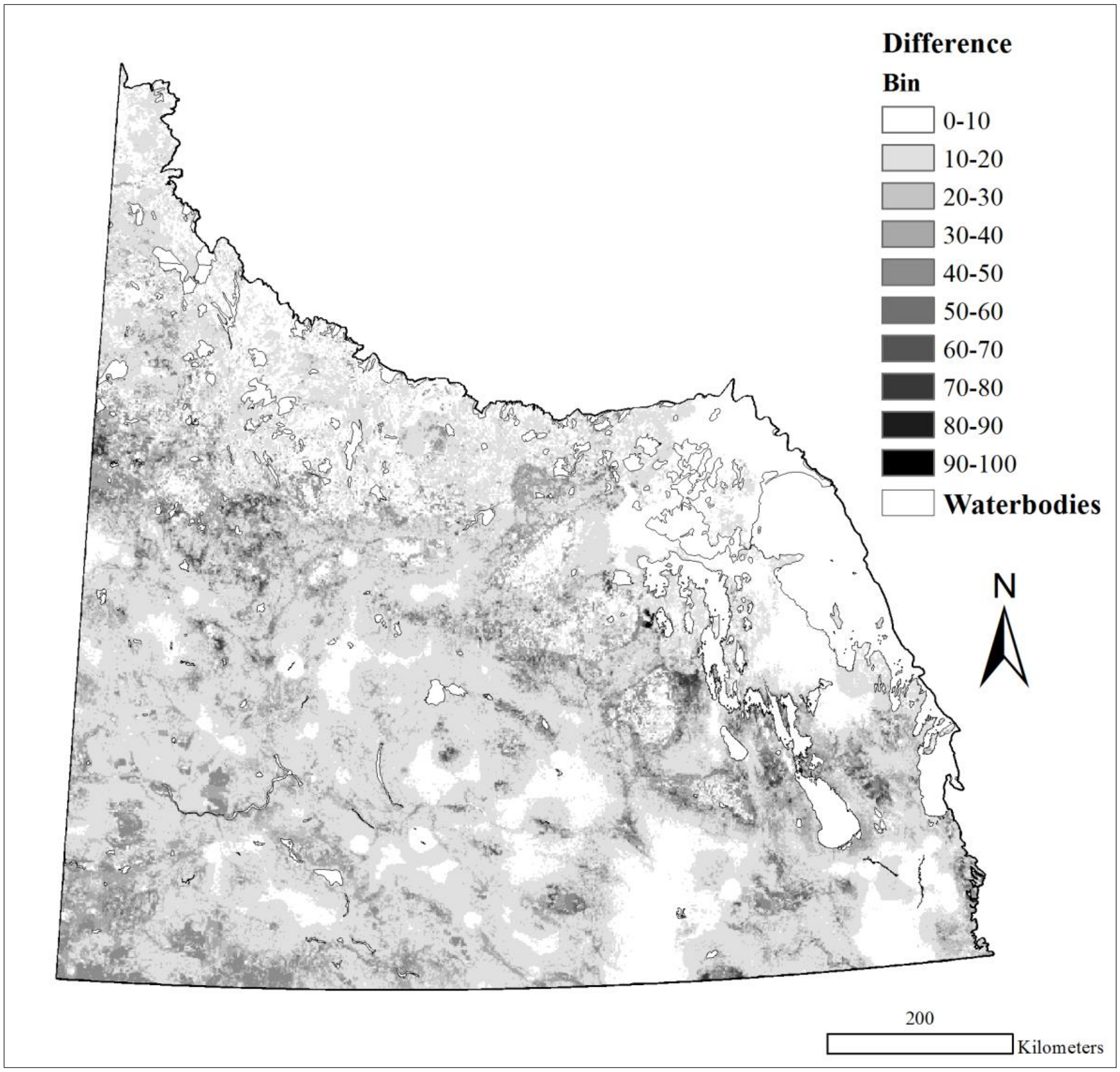

Figure 3.5 – RSF difference map between 2 RSFs depicting elk distribution in central Saskatchewan and Manitoba, Canada. Elk locations from two datasets were used to create the RSFs: a radio-collar dataset (*n* = 328 animals, 1998–2012) and local ecological knowledge (LEK) participatory mapping locations. This map shows the absolute difference between radio-collar data and LEK generated RSFs. Black indicates areas with the greatest absolute difference between RSFs, while low absolute difference areas are displayed as white or light grey. The bin designation is the percent difference between RSF values, where 100% is entirely different in that area, 50% is somewhat different and 0% is not different at all.

In contrast, in their assessment of distribution model performance, Johnson & Gillingham (2005) found that their expert-driven model was a poor predictor of species distribution, likely due to issues with experts recognizing which habitat is significant to the studied populations. By using participatory mapping as the method to gain LEK instead of asking experts to identify the important environmental factors, I was able to circumvent this potential issue. The method I developed also allowed the models to be evaluated equivalently but using empirical data, with recognition that neither knowledge system is more correct or superior to the other (Briggs 2005).

Resource selection patterns exhibited by elk were consistent with previous studies of elk in agricultural ecosystems. The variables that best predicted elk distribution — proximity to protected areas and paved road density — are related to elk security. In all three models, elk strongly avoided high densities of paved roads. Roads serve as a surrogate for human disturbance, and also allow access for hunters, thus areas with high road density are avoided by elk (Lyon 1979; Baasch et al. 2010; Proffitt et al. 2011). Elk selected for protected areas, which in this study indicates an area of native habitat with minimal human development. In many of these protected areas, hunting is allowed (Manitoba Conservation and Water Stewardship 2013; Saskatchewan Ministry of Environment 2013). The high degree of selection for protected areas may exist for several reasons. Elk may be seeking refuge from hunting by moving into protected areas since they alter habitat use in response to hunting (Burcham et al. 1999; Conner et al. 2001).

Reduced hunting risk may be an advantage to residing in protected areas, however, some areas that are classified as protected, such as some provincial parks and provincial forests, still allow hunting. Selection for protected areas may also occur simply because protected areas tend to be the largest patches of native habitat. Several protected areas within the study, most particularly, Riding Mountain National Park, function as an island of native forest within an agricultural-dominated landscape (Brook 2009). The extent to which hunting is responsible for the proximity to protected areas predictor variable importance is impossible to discern in this study since selection patterns are not separated into season. Additionally, elk resource selection differs by sex (McCorquodale 2003; Dugal et al. 2013). However, since the goal of this research is to determine elk distribution at a landscape scale, identifying seasonal and sexual selection patterns are not required to meet this objective.

The only discrepancy in selection patterns between datasets is the avoidance of coniferous forest in the LEK-derived model and the selection for coniferous forest in the collar and combined based models. The difference in selection patterns between datasets may reflect biases of both the LEK dataset and the radio-collar dataset. All three radio-collar datasets used in this study were recorded in habitat that is predominantly mixed wood forest (Ecological Stratification Working Group (Canada) 1996) with some coniferous forest. The typical habitat in these areas is quite different from that in the southern portion of the study area. Although elk require access to cover as escape from predators (Lyon 1979; Proffitt et al. 2011), elk in prairie habitats use other structures, such as groups of shrubs or topography, as cover in conjunction with trees (Bian & West 1997). Another study within the study area also found that elk avoid coniferous forest at the landscape scale (Dugal et al. 2013). Therefore, it is possible that elk selection as determined by the radio-collar and combined model overestimates selection for coniferous forest in some parts of the study area. In contrast, a possible limitation of the LEK collection process may underemphasize the importance of coniferous forest. While mapping, participants recorded locations where they saw elk or elk sign. In a dense coniferous forest, visibility is decreased and it may be more difficult to see elk or elk sign. As well, many locations identified by participants were from their experiences hunting elk. Hunters may avoid hunting in coniferous forest because it is more difficult to move through and access (Foster et al. 1997), which could bias the locations provided. As hunting becomes more difficult and prey density decreases, hunters expend less effort (Van Deelen & Etter 2003).

The survey results of factors important to elk do not indicate a general trend of agreement between participant opinions and actual selection patterns of elk. This result suggests that while participants know where they have seen elk, they do not always consciously evaluate the habitat types or environmental factors that exist around the elk sighting. It may be difficult for stakeholders to see the broader habitat characteristics that determine where elk are found. A study on hunting intentions found that the experience a hunter has is largely based on expectations for the event, not what actually occurs (Hrubes et al. 2001). This dissonance between intentions and experience could help explain these results. Hunters may associate a particular habitat type with elk, when in fact they actually find elk in a different habitat. The diversity of participants who replied to the survey may also account for the lack of congruency. Each group of wildlife users value different facets of their experiences with wildlife. For instance, previous research has shown that wildlife observers pay more attention to, and are more interested in learning about wildlife behavior than hunters or outdoor recreationalists (Daigle et al. 2002). In a prior study creating expert based distribution models, Johnson & Gillingham (2004) found that assessment of habitat importance variables, similar to the habitat importance survey used in this project, is sensitive to varying opinions. Participatory mapping does not require participants to consciously assess or analyze a wildlife experience; it is recording an observation. The participatory mapping derived locations may be less prone to observer bias than statements regarding elk habitat. Using different survey questions or researcher defined environmental variables instead of those defined by prior participants may also improve the congruency between survey results and quantitative assessment since the survey design can largely impact the quality of the responses received (Steele & Shackleton 2010).

Although all three RSFs were accurate when validated against an independent dataset, the LEK-derived RSF visually has more habitat categorized as a high likelihood of elk distribution than the collar-derived RSF. The process that created the LEK dataset may have overestimated elk distribution in the study area and thus selection, and the collar data may underestimate elk selection. This conclusion is supported by the analysis identifying the relative importance of each variable. Of the variable importance calculated for each dataset (w_+), 12 out of 14 are important ($w_+ > 0.5$) in the LEK dataset and 9 of 14 in the collar dataset. The spatial scale of the participatory mapping locations is much coarser than the radio-collar locations (the mapping data error was estimated to be 288 m, which is greater than the collar data). Radio-collar locations are points, while LEK locations are circled locations around a point, increasing the amount of habitat included as an elk presence point. The scale used in the maps provided to participants was quite large to allow most of the province to be printed on one sheet of paper. This scale also reduced accuracy. Moreover, the nature of the participatory mapping process, where there is potentially uncertainty due to participant memory and the digitization of the data, may be more error prone than the radio-collar data in certain situations. The collared animals in each radio-collar dataset are a subset of the elk population in that area. Even within the Riding Mountain dataset, which is the largest at 246 different animals, the collared animals only represent 12% of the entire population (given an average annual elk population of 2088 animals for the duration of the study (Brook 2008). The fewer collared elk in the population, the less likely rare habitat use patterns are going to be documented, consequently it is possible that the radio-collar datasets will not fully reflect the diversity of habitats occasionally used by elk that lie on the periphery of the animal's home range. Finally, the three collared elk populations are located in predominantly forested areas with close access to agricultural crops (Brook 2008; Hegel et al. 2009). For elk herds living in the forest fragments in the agriculture-dominated ecosystem of the south central portion of the study area, there is no radio-collar dataset. Thus, the radio-collar locations may underestimate the importance of agricultural crop, or grassland and not adequately reflect the habitat selection process for these elk.

While the results of this work are compelling and quantitatively accurate, there are several limitations of this research. First, the RSFs do not indicate where elk are located, but where the conditions are ideal for elk and thus are likely to be located. While the validation of the RSFs with the crop damage claims indicate that elk are present in many of the identified high quality habitat areas, without detailed population surveys of those areas it is impossible to say with certainty. Indeed there may be some areas where the RSF models (from either or both datasets) indicate a moderate or high probability of elk occurrence but that actually have little or even no elk. As well, this research looks at regional-scale elk distribution and selection patterns. Within each section identified as high quality habitat, hierarchical scales of selection processes are occurring (Johnson 1980). Elk may not be evenly dispersed within the identified high quality habitat areas, and at a smaller spatial scale, not all portions of sections identified, as high quality will actually be high quality.

Additional research is recommended to further validate the RSFs produced. New workshops where participants identify areas of accuracy and inaccuracy of the predictive models would assist the streamlining of the modelling process by highlighting strengths and weaknesses of the models created. Stakeholder viewpoints concerning which habitat types are important to elk would help identify why surveys did not correspond with quantitative estimates of environmental variables. As well, a more detailed social survey could help identify differences in how varying stakeholder groups perceive elk and elk habitat selection, which would assist any elk management plans. Finally, it would be valuable to follow the same procedure of mapping LEK with a different species to further evaluate this approach. Elk are moderate specialists when selecting resources, thus this method may perform differently with species that exhibit more or less specificity in their resource selection patterns (Mould & Robbins 1982; Wisdom & Thomas 1996).

This research affirms that LEK is an appropriate and useful tool in wildlife habitat selection studies and can be a viable, quantitative option for estimating distribution. LEK can be complementary to conventional expert-based research techniques and can be used to identify animal locations and habitat use patterns (Moller et al. 2004). Within this study, there were both direct and indirect benefits to using LEK. The greater spatial coverage of the local knowledge compared to the limited coverage of the radio-collar datasets allowed a better representation of the variety of elk habitat within the study area. Using LEK was also a less costly option. The estimated cost to obtain the radio-collar locations used in the study was $3 000 000 (Brook 2008; Hegel et al. 2009). In contrast, the total cost of collecting LEK for this study was $31 500. The LEK-derived model has an added advantage is that it is much more cost efficient and could be an effective option for wildlife researchers and managers limited by budget resources (Martin et al. 2012). Perhaps most importantly, the LEK gathering sessions facilitated stakeholder participation in elk management issues, as well as creating partnerships between workshop participants and researchers for future collaboration. Human values and beliefs are complex and dynamic, and often determine the success of a conservation effort (Lindenmayer & Hunter 2010). The use of LEK in wildlife research can contribute towards incorporating these beliefs, while resolving some limitations of technical biological datasets.

CHAPTER 4: IDENTIFYING CONSERVATION PRIORITIES FOR THE PERSISTENCE OF REMNANT CANADIAN PRAIRIE-PARKLAND ELK (*Cervus canadensis manitobensis*) POPULATIONS

4.1. Abstract

The anthropogenic fragmentation of the prairie ecosystem in North America has drastically reduced populations and ranges for many species. Of these, the prairie-parkland elk (*Cervus candensis manitobensis*) in Western Canada has recovered from near extirpation at the turn of the century, but remains vulnerable to a variety of potential threats, most importantly, continued habitat loss and disease. Within their historical range, elk populations remain clustered around forested protected areas and are not significantly re-established in their former prairie habitat. Currently, very little is known regarding the potential impact of these threats and how recovering populations may be affected. The goal of my research was to identify priority conservation areas to better inform prairie-parkland elk management. I used a previously developed resource selection function (Chapter 3) detailing elk distribution to identify irreplaceable areas of high quality habitat. I assessed the vulnerability of these areas by overlaying layers representing habitat loss from agriculture and forestry, disease and hunting. I evaluated the connectivity between these habitats and determined the contribution of currently established protected areas to elk populations. Overall, 5% (30 753 km^2) of the study area was classified as high quality (top 10% of RSF values) habitat encompassing 81 separate areas. Of these high quality patches of elk habitat, 87% had at least one vulnerability factor present. Core habitat areas endemic with chronic wasting disease were highly connected to many other high quality habitat areas. Elk populations continue to be tied to protected areas, with 88% of identified quality habitat core areas located inside protected areas. Given the overall lack of high quality habitat outside of protected areas, and the absence of protected areas in the interior, agricultural region of the study areas, elk management actions should focus on protecting current elk populations, as reestablishment outside of these areas seems unlikely. This research has shown that habitat loss and the spread of chronic wasting disease have a significant potential to negatively impact core populations areas. The long-term persistence of elk populations will be determined by the maintenance of current disease free, high quality habitat areas.

4.2. Introduction

Elk (*Cervus canadensis*) are an iconic species and important natural resource across North America. They are have a large impact ecologically, indirectly impacting the abundance and diversity of other species through their foraging activities, which affect nutrient cycling and plant community successional trajectories (Ripple & Larsen 2000; Kie et al. 2003). Elk are a key food source for many top predators, particularly wolves (Carbyn 1983) and are also hunted extensively, often as a traditional food source for First Nations people (McCabe 2002; Brook 2009). Elk are often regarded as a symbol of wilderness and provide non-consumptive services to wildlife watchers (Heydlauff et al. 2006). Prior to European settlement, elk were the most abundant cervid in North America and ranged widely across a variety of habitats (Bryant & Maser 1982; O'Gara & Dundas 2002). Unregulated hunting, in conjunction with habitat loss and modification, caused a drastic population collapse (Wisdom & Cook 2000; Laliberte & Ripple 2004). The prairie parkland or Manitoban elk (*Cervus canadensis manitobensis*) was extirpated across much of its range in the Great Plains (Soper 1946). The enactment of hunting regulations and the creation of parks, which functioned as refuges, allowed elk to regain some of their former range in the early 1900's, however very few animals remained on the prairies (Rivard et al. 2000; Brook 2009). Currently, populations continue to be clustered around protected areas, such as Riding Mountain National Park, Prince Albert National Park and Cypress Hills Interprovincial Park, but have expanded into the agricultural-dominated prairie region (Polziehn et al. 1998). Current population estimates for elk are 15 000 in Saskatchewan and 7350 in Manitoba, although the estimates for most areas are based on irregular and coarse grain surveys (Arsenault 2008; Manitoba Conservation 2013). Very little information exists regarding elk behaviour and abundance outside of protected areas and the relative contribution other habitat types make to population persistence.

Elk populations within the prairie-parkland region continue to face several threats to range expansion and population growth, particularly from disease and habitat loss. Two diseases endemic to the study area have the potential to impact elk populations – chronic wasting disease and bovine tuberculosis (Conner et al. 2008). Chronic wasting disease (CWD) is a fatal neurodegenerative disease that infects cervids and is currently present in North America in 18 states and two Canadian provinces, Alberta and

Saskatchewan (Williams et al. 2002; Tapscott 2011). Thus far, CWD has been found in elk, mule deer (*Odocoileus hemionus*), white tailed deer (*Odocoileus virginianus*) and moose (*Alces alces*) which all exist in the prairie and parkland region of Canada and exhibit considerable spatial overlap in distributions (Kahn et al. 2004; Conner et al. 2008). Currently, there is no evidence that CWD can be transmitted to humans or livestock (Raymond et al. 2000; Williams & Miller 2004; Saunders et al. 2012). In areas endemic with CWD, prevalence and spatial range of the disease are increasing. Once CWD is well established in a region, it can be difficult to eradicate due to a lack of feasible management options (Saunders et al. 2012). CWD is transmitted from direct animal contact and indirectly, from environmental contamination (Miller et al. 2006; Argue et al. 2007; Mathiason et al. 2009). Since 2008, CWD has been detected in 6 wild elk in Saskatchewan (Bollinger et al. 2014).

Bovine tuberculosis (bovine TB), caused by the bacteria *Mycobacterium bovis,* can infect many mammalian hosts worldwide (Daszak et al. 2000). In North America, some ungulate populations are thought to be wildlife reservoirs for the disease. Within the prairie-parkland region, elk in Riding Mountain National Park, Manitoba are suspected to be the primary wildlife reservoir for the disease (Lees 2004; Shury & Bergeson 2011). Bovine TB is spread from direct contact between animals as an aerosol (Francis 1958). Infection results in chronic respiratory disease (Renwick et al. 2007). The presence of bovine TB has the potential to indirectly hurt elk populations by causing massive population reduction, in an attempt to eradicate the disease and limit spread to livestock (Brook & McLachlan 2006; Brook 2009). To ensure healthy elk populations, currently unknown information regarding the nature of critical elk habitat, its distribution and abundance, and threats to that habitat need to be clarified (Cianfrani et al. 2010; Hebblewhite et al. 2012).

Conservation biology is an inherently crisis driven discipline, where decisions are reactionary in nature (Soulé 1985). As such, preventative research that establishes conservation priorities prior to a crisis can be beneficial as it directs the often limited monetary and workforce resources so that they can be used most effectively (Margules & Pressey 2000; Bottrill et al. 2008). In order to successfully conserve a species, the anthropogenic threats to their habitat need to be quantified (Fischer & Lindenmayer

2007). Therefore, to ensure the persistence of elk populations, current critical habitat, as well as future impacts on habitat need to be identified and assessed (Cianfrani et al. 2010). Overlaying measures of habitat irreplaceability and vulnerability to potential threats can assist in the prioritization of important areas (Pressey & Taffs 2001).

Given the overall lack of knowledge regarding elk within the study area, the purpose of this study was to quantify the status of elk habitat and define management priorities for prairie-parkland elk. I analyzed a previously developed model of elk distribution (see thesis Chapter 3) and compiled information on factors that make elk populations vulnerable to population reduction and regional extirpation. My objectives were to: (1) identify core areas of high quality, irreplaceable habitat and assess the vulnerability of these areas to habitat loss, disease and population reduction; (2) quantify habitat connectivity between high quality core areas and predict which core areas are at greatest risk of disease spread; (3) clarify the role protected areas play in determining high quality elk habitat by specifically determining a) the amount of protected areas that are high quality habitat, b) the relationship between the size of the protected area and habitat quality, and c) the relationship between habitat quality and protected area type; and (4) define areas of priority conservation concern using habitat quality and vulnerability, endemic disease locations and connectivity as criteria.

4.3. Methods

4.3.1. *Study Area*

The study area (Figure 4.1) for this project was determined based on the historical distribution of elk in the prairie-parkland region (Soper 1946; Bryant & Maser 1982; Polziehn et al. 1998). I examined 614 091 km^2, which roughly includes the southern half of Saskatchewan and Manitoba, and is defined by the Prairie and Boreal Plains ecozones. Land cover and human use vary greatly within the region. The southern portion of the study area is dominated by intensive agriculture and is a modified prairie ecosystem, with little remaining native prairie (Samson et al. 2004). Key crops include cereal and oilseed. The livestock industry also plays a large role in land use with beef cattle as the dominant form of production (Brook & McLachlan 2006; Rashford et al. 2011).

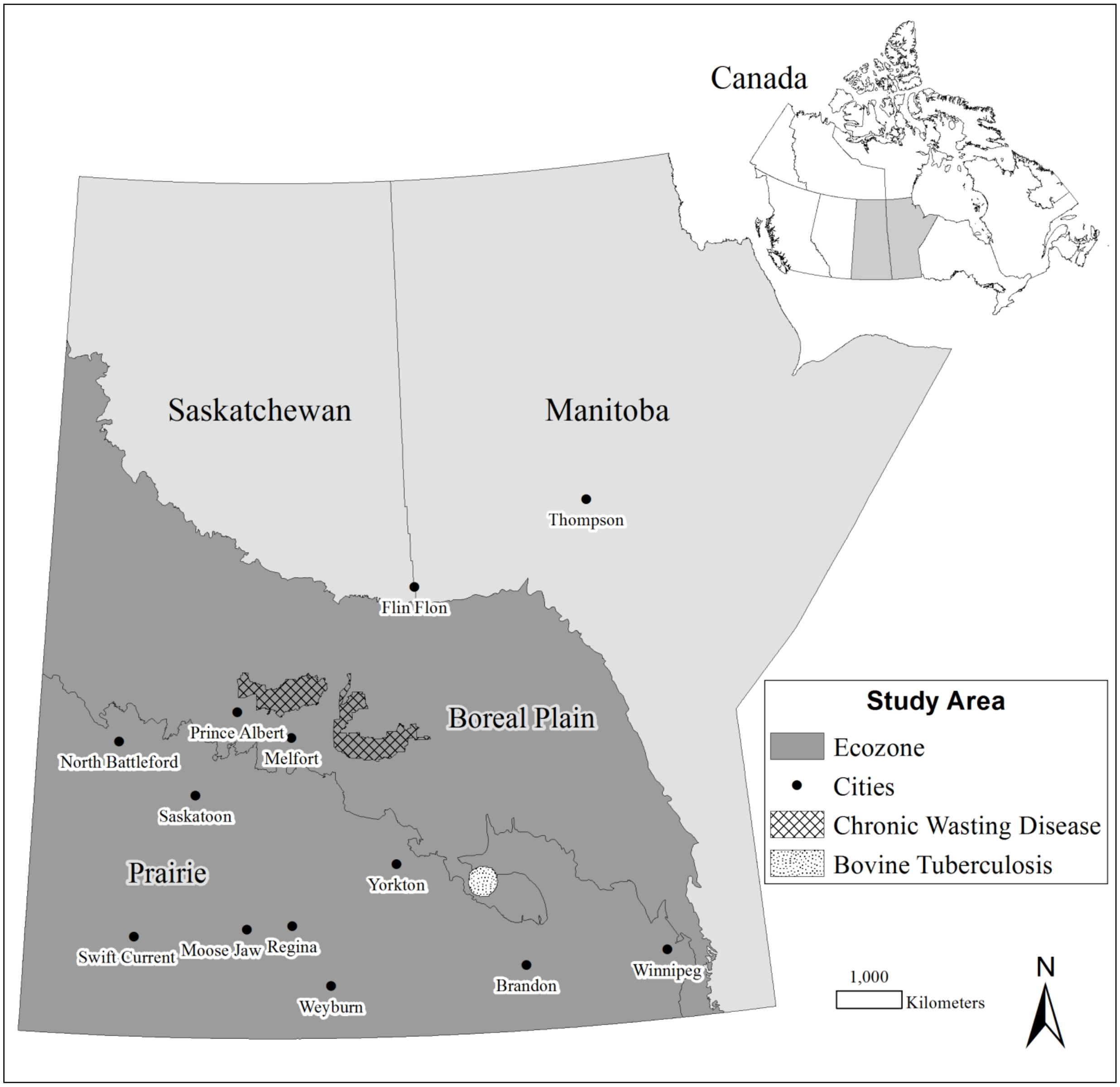

Figure 4.1 – Prairie and boreal plains ecozones, which comprise the study area in Saskatchewan and Manitoba, Canada. Major cities in the area are listed. Locations within the study area where wild elk populations *(Cervus canadensis manitobensis)* are endemic with chronic wasting disease and bovine tuberculosis are displayed.

The northern section of the study area is more deciduous and mixed forest dominated and consists of mixed wood boreal and deciduous forest with species such as trembling aspen (*Populus tremuloides*), jack pine (*Pinus banksiana),* balsam poplar *(Populus balsamifera)* and black spruce (*Picea mariana;* Ecological Stratification Working Group (Canada) 1996; Hobson et al. 2002). Agricultural land use and rural development continues to grow along the northern boundary of the forest, with an annual rate of forest to agricultural conversion of 0.75% from 1916 to 1996 (Bethke & Nudds 1995; Hobson et al. 2002). Increased deforestation and agricultural development have resulted in a modified ecosystem with habitat fragmentation, wetlands and bogs loss, and a change in fire regime (Bethke & Nudds 1995; Hobson et al. 2002; Fitzsimmons 2003; Samson et al. 2004).

4.3.2. *Resource selection function distribution model*

I used a previously developed resource selection function (RSF) model (see Chapter 3 for more details) to identify high quality elk habitat across the study area. An RSF is a predictive model of the relative probability of animal occurrence based on a set of environmental predictor variables, and derived using logistic regression (Table 4.1) (Manly et al. 2002). RSFs can be used as a standalone tool to assess selection and occurrence patterns but can also be applied to additional analyses involving animal movements and landscape connectivity (Chetkiewicz & Boyce 2009). I created an RSF depicting elk distribution in the study area using elk locations from radio-collar datasets and a local ecological knowledge derived mapping dataset. Collar locations were collected in Cypress Hills (SK/AB, n = 64 VHF collared females), Montreal Lake (SK, n = 18 VHF collared females), and the Riding Mountain Region (MB, n = 212 VHF collared (120 females and 92 males) and 34 GPS collared (24 females and 10 males)). The Cypress Hills dataset was collected from 1998 to 2000 and Montreal Lake from 1999 to 2000. In the Riding Mountain Region (RMNP), I used locations collected between 2002–2012. To gain participatory mapping data, I held 10 knowledge-gathering workshops, where 71 local experts (individuals who had experiential and/or academic about elk herds in their area) annotated a landcover map with locations where elk herds were consistently, frequently and regularly located. A total of 392 elk locations were identified and digitized using Arc10 (ESRI 2011). I choose environmental predictor covariates based on attributes known to influence elk distribution (Sawyer et al. 2007; Brook 2010; Baasch et al. 2010; Dugal et al. 2013). I evaluated elk selection using logistic regression at the landscape scale with the program R (R Development Core Team 2011), with sections being the unit of measurement. I then mapped the probability of elk use across the study area in Arc10 (ESRI 2011).

Table 4.1 – Environmental predictor variables and their descriptions as used in a RSF model depicting elk distribution across central Saskatchewan and Manitoba. Elk locations were from a dataset including radio-collar data (*n* = 328, 1998–2012) and local knowledge participatory mapping locations. The estimated β coefficients and standard error as calculated in the RSF are shown. Landcover descriptions are adapted from Wulder & Nelson (2003).

Environmental Predictor Variable	**Layer Description**	***β* Coefficient Estimate**	**Standard Error**
Coniferous Forest	% of section containing coniferous forest where 75% of the basal area is covered by coniferous trees such as jack pine or black spruce, 10–100% crown closure	0.49	0.16
Crop	% of section containing annual agricultural cereal, pulse, and oilseed crops	1.35	0.11
Deciduous Forest	% of section containing coniferous forest where 75% of the basal area is covered by deciduous trees such as trembling aspen and balsam poplar, 10–100% crown closure	2.66	0.13
Distance to Park	Mean distance per section to provincial and national parks and provincial forests	-6.47	0.33
Forage	% of section containing perennial cropland and pasture	2.01	0.17
Grassland	% % of section containing mixed native and tame grasses and herbs with <10% shrub cover	1.27	0.14
Herb	% of section with greater than 20% cover of vascular plants without a woody stem such as grasses or forbs	2.86	0.39
Mixed Wood Forest	% of section where no more than 75% of the area is classified as either coniferous or deciduous forest, 20–100% crown closure	2.56	0.16
Paved Road Density	Mean density of paved roads with a township per section	-25.58	3.63

4.3.3. *Core areas of high quality elk habitat*

I determined where core elk habitat was located within the study area using the previously created RSF as a measure of habitat quality or irreplaceability. The irreplaceability of an area is the value that area has as wildlife habitat, or in other words, the contribution it makes towards a conservation goal (Pressey & Taffs 2001; Carroll et al. 2003). I defined high quality elk habitat as areas with an RSF value in the upper tenth percentile of the RSF value distribution (Carroll et al. 2003; Chetkiewicz & Boyce 2009). The high quality areas identified needed to be capable of supporting an elk herd, thus I only included areas that were greater than 50 km^2 in size. This number was based on previously identified elk home range size (Anderson et al. 2005a).

4.3.4. *Threatened core area identification*

Elk populations within the study area are currently undergoing several threats to population persistence, including habitat loss and disruption due to forestry and conversion to agriculture; disease; and hunting. The combined impact of these factors could be significant enough to drive elk populations downward (Fischer & Lindenmayer 2007). Assessing the vulnerability of an area to these factors will assist conservation planning. In this context,vulnerability is the risk that an area of habitat will be modified or harmed by extractive industries or human use (Margules & Pressey 2000).

Forestry can be problematic for elk due to habitat modification where the forest changes from closed canopy mature forest to open clear-cut forest since elk require forest cover (Boyce et al. 2003; Sawyer et al. 2007). Forestry is also a disturbance, which is detrimental to fitness (Frid & Dill 2002; Naylor et al. 2009) and causes increased human access and road creation in elk habitat, which alters habitat use patterns and increases hunter accessibility (Lyon 1979, 1983; Unsworth et al. 1998; Creel et al. 2005) . Within Saskatchewan, forestry has decreased the size of native vegetation patches and reduced ecological functioning (Fitzsimmons 2003). While elk do use clear-cut areas, they prefer more dense forest cover (Davis 1977). Conversion of native habitat to agriculture is a main driver in worldwide species extinction and biodiversity loss (Vitousek et al. 1997). In North America, grasslands have been intensely modified as a result of agricultural conversion (White et al. 2000). Conversion to agriculture specifically affects elk by reducing the availability of appropriate habitat year round, because areas that are best for agriculture are often elk winter refuges (Cole 1971; Vavra 2006). In addition to habitat loss, conversion to agriculture of former elk habitat can also result in human-wildlife conflict, which negatively impacts elk (Walter et al. 2010).

Bovine tuberculosis and chronic wasting disease are two diseases found within the study area (Conner et al. 2008). The spread and prevalence of bovine TB has the potential to directly negatively affect elk populations through disease impacts on animal survival, but more importantly, can cause massive population reduction by humans as a disease eradication strategy (Brook 2008). CWD poses a major threat to elk populations in the prairie-parkland due to the ability of the infectious agent to persist in the environment for multiple years (Williams & Miller 2004), the overlapping range host species in the region, and the lack of an existing prophylactic (Bollinger et al. 2004; Conner et al. 2008; Saunders et al. 2012). Even when animals are harvested at sustainable levels, hunting can have negative consequences for elk. Hunting has been shown to influence elk movement patterns, habitat selection, reproductive success and foraging routines (Phillips & Alldredge 2000; Conner et al. 2001; Christianson & Creel 2007; Proffitt et al. 2010; Ciuti et al. 2012). Additionally, hunting can serve as a disturbance, causing taking energy away from other activities that would increase fitness (Conover 2001; Frid & Dill 2002). Thus, in addition to number reduction, hunting, and the increased vigilance it causes, may have a cumulative negative impact on elk populations.

Forestry is allowed in all provincial forests in both Saskatchewan and Manitoba (Government of Manitoba 1988; Government of Saskatchewan 1996). I used GIS layers depicting all provincial forests to represent the presence of forestry. While some areas with the provincial forest designation are not currently harvested, in the interest of long-term conservation planning, all provincial forests were included on the analysis. Agriculture and Agrifood Canada's soil database and land inventory was used (Agriculture and Agri-food Canada 1998) to identify areas at risk of agricultural conversion. I designated areas with a soil classification of classes 1-4 (areas that have no significant limitations for crops to areas that have severe limitations on crop types) as areas that are at risk. To create spatial layers representing areas endemic to disease, I used a previously identified hotspot for bovine tuberculosis outbreaks (Shury, unpublished data) and the reported locations of 6 wild CWD positive elk (Canadian Cooperative Wildlife Health Centre, 2013). The wildlife management zones (49 and 50) where CWD positive elk have been found were used to denote locations endemic for CWD. Wildlife management zones are the unit used for all wildlife management decisions. Given that CWD sampling efforts have been limited due to declining hunter participation (Bollinger et al. 2014), it is a reasonable assumption that CWD is present in the wildlife management zone. I referenced the 2013 hunting guides for both Saskatchewan and Manitoba (Manitoba Conservation and Water Stewardship 2013; Saskatchewan Ministry of Environment 2013) to identify areas where elk hunting was allowed. Forestry and conversion to agriculture cause habitat loss and modification, while disease and hunting impact population numbers.

The layers representing all four vulnerability factors were overlaid with the identified high quality habitat in GIS (ESRI 2011). The layer depicting vulnerability to forestry covered the northern border of the study area and Saskatchewan with some coverage in the Western portion of Manitoba. The risk of agricultural conversion layer patchily covered the entire study area excluding some protected areas and did not meet the northern border of the study area in Saskatchewan. The disease locations used are visible in Figure 4.1. The hunting layer visually encompassed 60% of the entire study area and was most consistent in the north central portion of the study area. I combined these layers and identified which portions of the identified high quality habitat areas were covered by 0 – 4 of these factors.

4.3.5 *Functional landscape connectivity analyses*

To assess functional landscape connectivity between high quality habitat areas, I used a least cost path analysis (Chetkiewicz & Boyce 2009). A least cost path uses a measure of cost associated with movement between patches, which is assigned based on the attributes of an area. Attributes that facilitate movement have a low cost, and areas with attributes that hinder or prevent movement are given a high value. The path of lowest cumulative resistance is calculated between patches (Adriaensen et al. 2003).

I used the identified high quality habitat areas as source patches and the inverse of the RSF generated previously as a cost surface (Chetkiewicz & Boyce 2009). Least cost paths have been criticized for using impossible or improbable path lengths that are not grounded in the ecology and behavior of the study species (Sawyer et al. 2011). Therefore, I chose to create four least cost path dispersal scenarios based on documented radio-collared elk movements. The shorter distances were chosen to reflect highly probable annual movements, while the longer distances were chosen to demonstrate rare but possible elk movements (Armbuster unpublished data, 2003; Brook 2008; Dugal 2012). Additionally, I trimmed the width of the corridors to a maximum width of 10 km to reflect more biologically accurate movement patterns (Sawyer et al. 2011). Using Linkage Mapper software (McRae and Kavanagh, 2011), I ran four least cost path scenarios between the identified priority conservation areas based on documented elk movements (11 km, 22 km, 118 km and 380 km), where each scenario had a maximum cost weighted path length of the dispersal distance.

4.3.6 Identification of priority conservation areas

To identify potential priority areas for conservation efforts, I used the high quality habitat identified as most at risk of being impacted by the vulnerability factors. I defined highest priority conservation areas as those with three or four vulnerability factors present. Additionally, all priority areas were connected to disease endemic areas in one of the annual movement scenarios.

4.3.7 Assessment of the impact of protected areas

In the RSF used to determine high quality habitat areas, the environmental covariate, “Distance to park” influenced elk selection patterns the greatest, following “Paved road density” (see Chapter 3). I used the IUCN’s definition of protected areas which is “ an area of land and/or sea especially dedicated to the protected and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means” (World Conservation Union 1994). Within this variable, I included every category of land use protected from extensive human development, such as provincial, national and regional parks, wilderness areas, wildlife refuges, provincial forests and PFRA community pastures. Hunting is allowed in the majority of these protected areas with a few individual exceptions and is not allowed by members of the public in National Parks. The land use permitted in each type of protected area also strongly differs. For instance, forestry is allowed in provincial forests and many provincial parks, while the primary use of PFRAs is cattle pasture. National parks are the most restrictive in their permitted activities, with minimal land use outside of tourist infrastructure development allowed. Protected areas play a large role in conservation efforts and exist to preserve biodiversity and provide habitat for species (Gaston et al. 2008). However, in the case of cervids, areas surrounding protected areas and parks are often hotbeds of human-wildlife conflict, which can reduce the effectiveness of protected areas as a conservation tool (Madden 2004; Naughton Treves 2008). The region surrounding both Cypress Hills Interprovincial Park and Riding Mountain National Park have extensive histories of conflict between producers and elk (Brook 2008; Hegel et al. 2009).

To identify the relationship between protected areas and elk populations, I used GIS to compare the locations of protected areas and the core high quality habitat identified earlier in the analysis. I calculated the total area for the four main types of protected areas; provincial forests, provincial parks, national parks and PFRA community pastures. I then calculated the total amount of protected areas identified as high quality habitat out of the total size of protected areas across the study area and the total area of the high quality habitat core areas identified in core areas. For the remaining analyses, I dropped national parks because there were only two in the study area. I ran a logistic regression on the statistical program R (R Development Core Team 2011) to determine the relationship between protected area size and habitat value. To establish if habitat quality is determined by protected area type, I ran a Kruskal -Wallis one-way analysis of variance in R (R Development Core Team 2011) with RSF values for provincial and regional parks, provincial forests and PFRA community pastures.

4.4. Results

4.4.1. *High quality habitat identification*

Using the RSF previously produced, I identified 81 core areas of high quality elk habitat (top 10% of RSF values) that were greater than 50 km^2 in total size. The total area of high quality habitat was 30753 km^2 or 5 % of the total study area. High quality habitat areas were largely located in the northern half of the study area along the forest fringe (Figure

4.2). The mean area of the identified sites was 371 km^2, while the minimum and maximum areas were 51 km^2 and 5 066 km^2 respectively.

4.4.2. *Threatened core areas*

Of the identified high quality habitat areas, approximately 87% of identified high has at least one vulnerability factor present (Table 4.2). The forestry layer covered 45% of identified core areas, the agricultural conversion layer covered 26%, the hunting layer covered 63% and the disease layer covered 6%. The majority of habitat core areas had two vulnerability factors present. Only four identified core areas were completed unaffected by any vulnerability factors.

Table 4.2 – Total area of core area of high quality elk habitat in central Saskatchewan and Manitoba, Canada covered by 1–4 of the following: hunting, disease (CWD or bovine tuberculosis), forestry and high risk of conversion to agriculture.

# of Vulnerability Factors	Area Covered (km^2)	% of Total Core Habitat
1	5 530.5	18.0%
2	12 190.0	39.6%
3	8 884.0	28.9%
4	94.6	0.3%
Total Area Core Habitat:	30 753.0	86.8%

4.4.3. *Connectivity scenarios*

The first dispersal scenario of 11 km resulted in 16 isolated core areas and five distinct clusters where neighboring core areas were connected (Figure 4.3). A possible movement of 22 km yielded 16 unconnected core areas and 3 clusters of connected core areas. A dispersal of 118 km had one cluster of connected core areas and one unconnected, isolated core area, while a possible dispersal of 380 km resulted in no isolated core areas and one interconnected cluster of all core areas.

The core areas with CWD present were interconnected with other areas in all dispersal scenarios. The core area containing bovine TB was not connected to other core areas until a potential dispersal distance of 118 was used. The area with endemic bovine TB continued to be connected when I used a dispersal scenario of 380 km.

4.4.4 *Priority conservation areas analysis*

Of the 81 areas of core habitat, portions of 40 were designated as priority areas for conservation action based on the number of vulnerability factors present and their proximity to core areas with CWD (Figure 4.4). All areas identified as a priority for conservation were located in the northern portion of the study area. The total area given the priority designation was 8 865.4 km^2 or 29.0% of the total high quality habitat area identified.

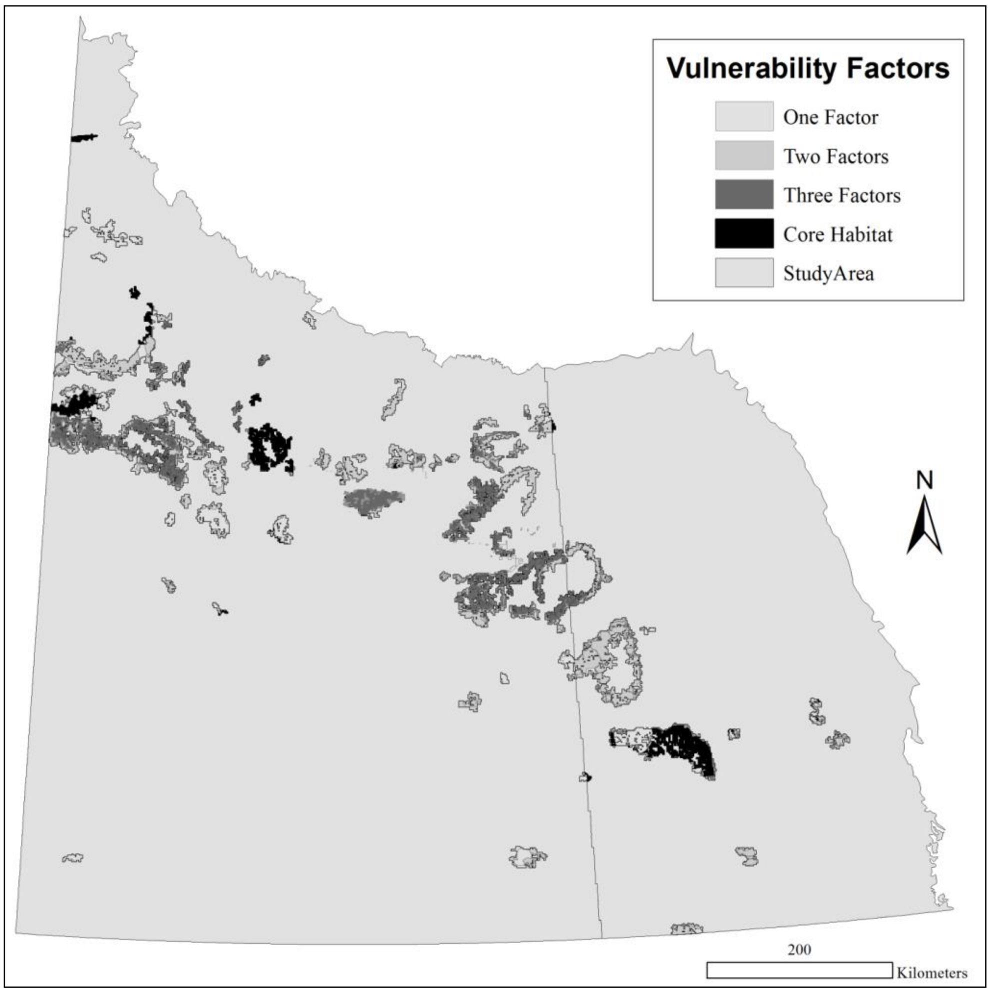

Figure 4.2 – Identified 81 areas of high quality elk habitat within central Saskatchewan and Manitoba, Canada. Core areas that contain 0–3 of the following: hunting, disease (CWD or bovine tuberculosis), forestry and high risk of conversion to agriculture are shown.

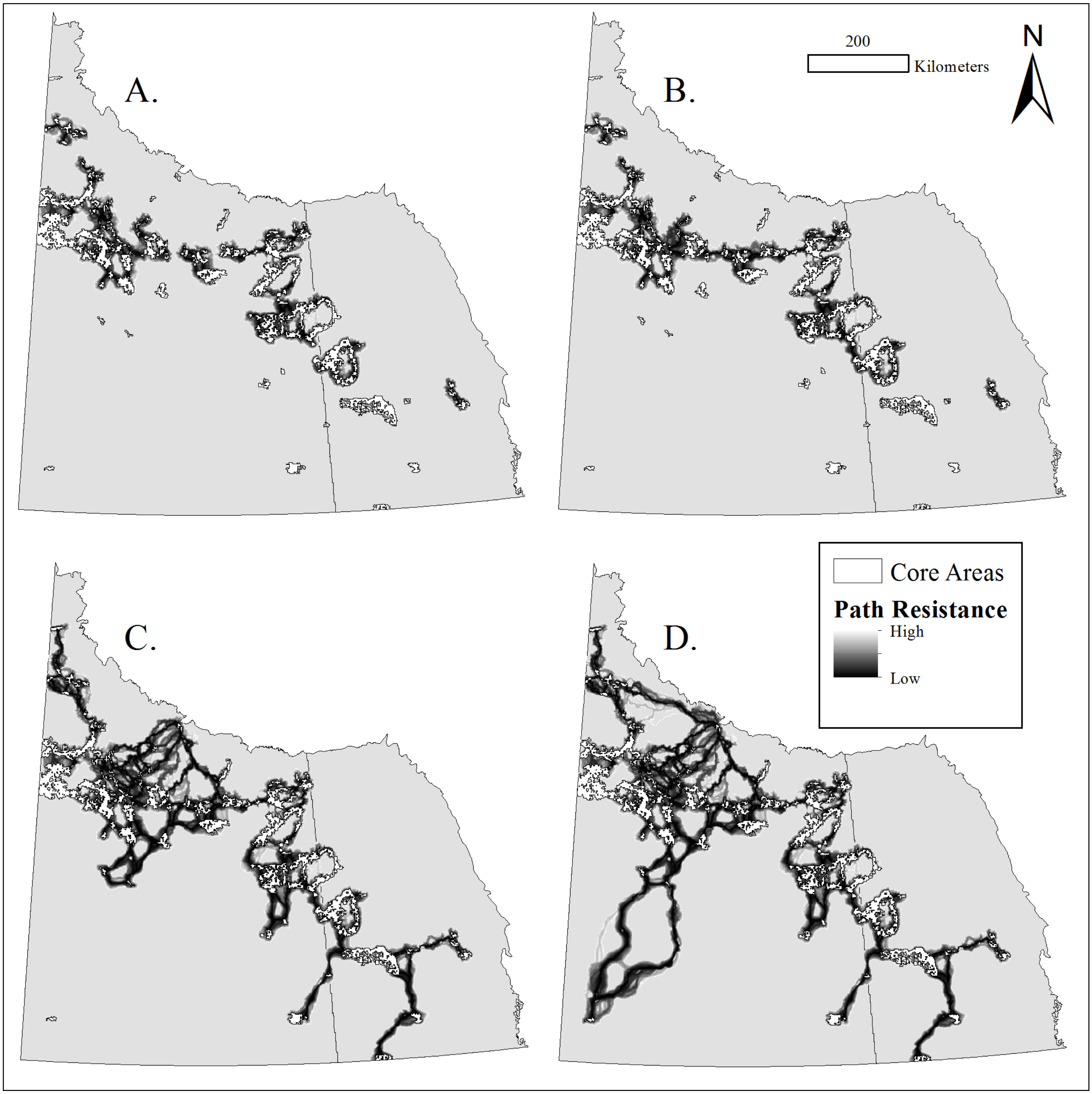

Figure 4.3 – Maps showing least cost paths between 81 core areas of high quality elk habitat in central Saskatchewan and Manitoba, Canada with four maximum dispersal scenarios: (A) 11 km, (B) 22 km, (C) 118 km and (D) 380 km and a maximum path width of 10 km.

Figure 4.4 – High quality areas of elk habitat in central Saskatchewan and Manitoba, Canada identified as most vulnerable based on the cumulative number of the following factors: hunting, disease (CWD or bovine tuberculosis), forestry and high risk of conversion to agriculture, as well as proximity to disease endemic areas.

4.4.5 Protected area analysis

Of the identified core habitat areas, 3748.6 km^2 or 12.2% were located outside of protected areas. The total amount of protected areas in the study area is 69 620.2 km2, with the majority in provincial parks (Figure 4.5). Of the protected areas within the study area, 5% are categorized as a core area of high quality habitat. The habitat outside of protected areas existed along the outer edge of the core areas. No entire core area or interior portion of a core area was located outside of a protected area. There was no relationship between the size of a protected area and the RSF value ($r^2 = 0.0002$, p = 0.67). The mean ranks of the RSF values are not significantly different (chi-squared = 3.53, p = 0.17) among the three categories of protected areas (provincial park, provincial forest and PFRA pasture; Figure 4.6).

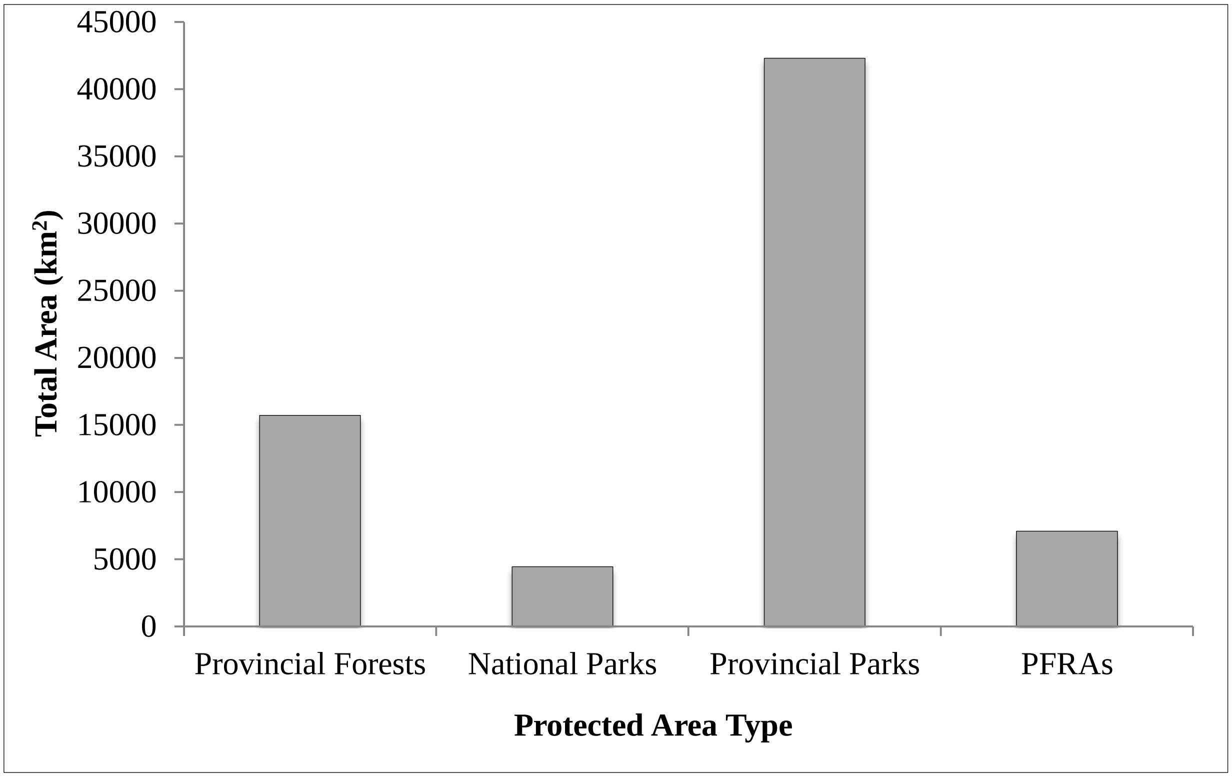

Figure 4.5 – Total area in kilometers squared of the four major protected area types in central Saskatchewan and Manitoba, Canada.

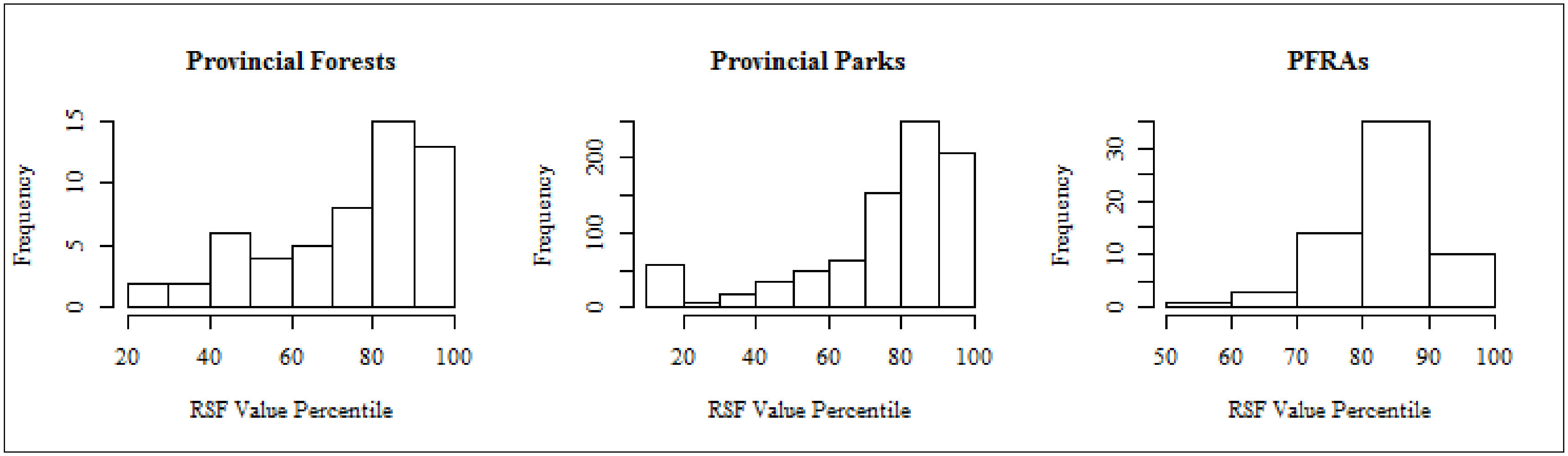

Figure 4.6 – Allocation of RSF values from an elk distribution model in central Saskatchewan and Manitoba in each spatial unit for three categories of protected areas: Provincial forests, provincial parks and PFRA community pastures. Values from 90–100 are the top 10% highest RSF values. The higher RSF value, the higher likelihood elk will be located in that area because of the combination of habitat attributes. Elk locations used to create the RSF were from dataset including radio-collar data (*n* = 328, 1998–2012) and local knowledge participatory mapping locations.

4.5. Discussion

My research successfully identified high quality core areas of elk habitat vulnerable to human influence, habitat loss and disease. The lack of core elk habitat in the southern portion of the study area indicates that elk are unlikely to re-establish prairie range lost at the turn of the century. Connectivity between many of the core habitat areas has the potential to ensure population growth; however, it may also facilitate the spread of CWD between core elk populations. Core areas of habitat are all closely related to protected areas, which may be the result of the amount of native habitat present in certain protected areas. Managing elk populations for persistence should focus on reducing future habitat loss and minimizing CWD spread in core habitat areas.

Effectively managing species for conservation requires explicit goals and objectives, and needs to take place within a decision-making framework (Margules & Pressey 2000; Lindenmayer & Hunter 2010). My work to identify priority conservation areas for elk is a vital first step in this process (Pressey & Taffs 2001). Elk populations are concentrated in and around protected areas, which can make the conservation process more difficult and prone to conflict (Madden 2004). Managing species in and around protected areas requires balancing human use, such as hunting and forestry, with species priorities to ensure that ecological function is being preserved (Gaston et al. 2008; Lindenmayer & Hunter 2010). DeFries et al. (2007) define three objectives that facilitate successful conservation planning when dealing with protected areas. The first is the identification of attributes of concern, in this case, elk populations. The second objective is to determine landscape connectivity to identify locations where protected area function is dependent on areas of the landscape that are unprotected. The third is to establish and take into consideration the socio-economic perspectives from users of that area that determine use of land resources. I was able to define the first two requirements; however, further research identifying perspectives of elk users such as hunters, outfitters, foresters and producers would be a valuable next step when planning for long-term elk management around protected areas. Additional next steps for elk population management across the study area include: a review of the status of individual populations in each core area, policy changes that support conservation actions which improve the functioning of elk core areas, and monitoring of core area habitat and populations after these changes (Margules & Pressey 2000).

Core habitat areas are clustered in the northern edge of prairie-parkland elk range. While elk do use agricultural ecosystems, they prefer areas that also provide access to considerable levels of cover. Previous studies on elk habitat selection patterns in the study area indicate that protected areas and the transition zone around them into the farmland provides elk with easy access to forage, as well as an escape route from predators and hunters (Lyon 1979; Brook 2010; Proffitt et al. 2010; Baasch et al. 2010; Dugal et al. 2013). Elk living in prairie ecosystems can use other landscape features such as topography and shrubby areas, as cover (Bian & West 1997); however, it is possible that the elk in the study area are better adapted to a forested environment, specifically when risk from hunters and anthropogenic disturbance is high. Additionally, elk may not make use of these structures since the prairie-parkland elk subspecies is not a prairie obligate and likely expanded its range into the prairies from the surrounding forests (Samson & Knopf 1996). Even prairie vegetation which appears unaltered by agriculture, is likely part of a different ecosystem than that of the pre-settlement prairie due to changes in large herbivore grazing and fire frequency (Samson & Knopf 1996). Therefore, it is likely that prairie-parkland elk are better adapted to areas with ample access to forested cover than a human-modified landscape with reduced cover. The lack of core areas outside of large protected areas in the south indicates that there likely is not enough contiguous native habitat for large numbers of elk to re-establish themselves in the central portion of their range.

Past research has shown that the cumulative effect of deterministic threats to population persistence, such as decreases in native vegetation from forestry (Gilpin & Soulé 1986), can make a species more vulnerable to stochastic events such as a natural disaster or disease outbreak (Fischer & Lindenmayer 2007). Overall, the probability of extinction is directly related to any habitat changes (Fahrig 2003; Fischer & Lindenmayer 2007). There is evidence that the interactions between these factors can exacerbate the negative effects caused by each other. For example, sociality can interact with hunting to increase disease prevalence, which can in turn increase the probability of species extirpation, even though hunting on its own is not problematic (Choisy & Rohani 2006; Donnelly et al. 2007). Although hunting was allowed in over half of the core areas identified in this research, it is most likely not a concern for elk populations on its own. In elk populations across North America, hunting and poaching are main causes of mortality (Stussy et al. 1994; Ballard et al. 2000; Frair et al. 2007). However, if populations are otherwise stable, hunting can occur without harming population health (McCorquodale 1997;

Hessl 2002). While the four factors used to assess elk extinction vulnerability in this study may not have an equal effect on elk population reduction, establishing where these factors overlap, and their potential to interact with each other and with human dimensions, allows managers to better focus conservation efforts.

Thus far, no regional scale connectivity analysis has been done on elk populations in the study area outside of the Riding Mountain and Duck Mountain region. While the connectivity results in this paper are at a coarse scale, the results do provide a baseline to estimate movement patterns. The RSF used to create to connectivity analysis was validated, but the identified corridors were not. Without landscape scale elk movement data throughout the study area, it is difficult to assess the accuracy of this analysis. However, the corridor identified by the two long distance scenarios corresponds to a previous connectivity study between Duck Mountain and Riding Mountain, which does provide a rough form of validation (Dugal 2012). Furthermore, given the extreme risk of disease becoming endemic in new areas, it seems prudent to consider the potential for longer scale dispersal. Connectivity between elk in core areas has the potential to improve population viability by allowing animals to move between patches, and rescue subpopulations from inbreeding and stochastic threats (Gilpin & Soulé 1986; Hanski 1998). However, increased connectivity, especially for the core areas connected to endemic CWD positive areas with likely annual movement disctances, has the potential to facilitate the spread of CWD, which may negate any benefits from metapopulation reestablishment. Disease epidemiology mirrors the dynamics of metapopulations, where habitat fragmentation can make disease transmission more complex (Grenfell & Harwood 1997). The impact of habitat patch size and fragmentation extent on disease transmission and persistence is difficult to quantify, and highly dependent on specific epidemiology measures. It is possible that the intermediate sizes of many core habitat areas may facilitate CWD persistence (Etienne & Heesterbeek 2000).

The protected area analysis indicated that protected areas are important to elk, but are not the only factor in determining high quality habitat, as many protected areas in the southern portion of the study area were not identified as core elk habitat. The absence of a relationship between protected area type and size, and habitat quality was unexpected. Protected area size is correlated with human density, where smaller protected areas exist in areas with a higher anthropogenic footprint (Parks & Harcourt 2002). Given this phenomenon, I would expect smaller protected areas to have lower quality habitat and larger areas to have higher quality habitat, but this is not the case. This discrepancy may be explained by the extreme variation in protected area size. To determine the RSF value, I calculated the mean value per area. It is possible that very large protected areas contain a large degree of habitat variation and different RSF values, which may have resulted in a lower overall RSF value. Previous research has shown that use of protected areas by wildlife is dependent on land use practices outside of the park (Rivard et al. 2000). Regional extirpations are heavily linked to characteristics surrounding the park, rather than the quality and size of the park itself (Rivard et al. 2000). Parks in the southern portion of the area are surrounded by more intensive land use practices, which may hinder the settlement of these areas by elk unless the protected area has additional attributes that compensate for this issue. It is also possible that there is an interaction effect between size and another factor because protected areas in the central portion of the study area where no core areas identified are generally smaller, while protected areas in the northern portion of the study area are larger. Additional analyses on the influence of protected area size on habitat quality would be beneficial.

Regarding protected area type, it is possible that the classification of being protected does not matter, but the type of habitat within the protected area does. Protected areas have been shown to adequately protect forest ecosystems but protect other ecosystems to a lesser extent (Geldmann et al. 2013). Typically, existing protected areas represent a biased sample of possible habitat types as they tend to be created in locations that are viewed as less valuable for extractive or consumptive human use, and are often selected primarily for human recreation purposes (Margules & Pressey 2000). Protected areas in the Saskatchewan prairie unequally represent habitat types and were established based on the suitability of the area for agriculture, not the quality of habitat for wildlife (Fore et al. 2013). In the south portion of the study area, protected areas may not be containing habitat required by elk and conversely, may not be protecting the most important areas. Protected areas may be important to elk simply because they contain native habitat (Fischer & Lindenmayer 2007). However, as the relationship between size and RSF value was not significant, the amount of remnant native habitat alone cannot fully explain this trend. Finally, it is possible that current population distribution are still somewhat determined by the locations where elk recovered in at the turn of the century, namely Cypress Hills Interprovincial Park, Prince Alberta National Park and Riding Mountain National Park (Soper 1946; Brook 2009), and have not been able to readily disperse from these areas. Overall, determining why protected areas are important indicators of elk presence requires more research before a conclusion can be made with certainty.

Although this paper uses disease locations, it does not include an epidemiological assessment of how CWD and bovine tuberculosis may move through elk populations. While an epidemiological assessment may be possible for the well-studied bovine tuberculosis system in Riding Mountain, Manitoba (Lees 2004; Shury & Bergeson 2011; Brook et al. 2013), the lack of data regarding CWD transmission in wild elk in Saskatchewan, and a sampling effort that is limited by a now defunct hunter participation program, makes it difficult to estimate current prevalence and potential infection rates for CWD (Rees et al. 2012; Bollinger et al. 2014). Transmission of disease depends on animal sociality and genetic relatedness, in addition to landscape factors (Farnsworth et al. 2005; Blanchong et al. 2008; Grear et al. 2010). Previous work on the transmission of bovine TB in the study area identified transmission as density dependent (Vander Wal et al. 2012). It is unlikely that bovine TB would spread outside of the endemic subpopulation because of the relative social isolation of those groups (Vander Wal et al. 2012). Coupled with the connectivity analysis performed in this paper, which demonstrates how the bovine TB endemic area is spatially isolated, it is very unlikely that bovine TB will be a threat for elk populations in the study area outside of the immediate endemic zone. While social mechanisms such as relatedness may limit the spread of CWD in direct animal to animal contact (Grear et al. 2010), the persistence of the infectious agent in the environment limits the isolating effect (Saunders et al. 2012). Additional research and disease monitoring is required to further predict how CWD may move through elk populations.

Although the RSF used to define quality habitat core areas was accurate, as determined by validation with an independent dataset (Boyce et al. 2002; Austin 2007), I was unable to obtain population estimates for specific regions across this study area. Therefore, I was unable to determine current elk occupancy of core areas and relate habitat quality to density. Relating population density to habitat quality would validate the selection of important high quality habitat (Boyce & McDonald 1999; Aldridge & Boyce 2007). Establishing population baselines for elk populations in each core area would also allow better management, as the health of each metapopulation would be able to be better determined and monitored. Additionally, linking population density to habitat quality may allow selection of other areas of important elk habitat that the analysis overlooked.

This research is the first to demonstrate that elk populations throughout the prairie-parkland remain vulnerable to population reduction. Elk have recovered from a previous range collapse and regional extirpation, but core areas of habitat continue to be impacted by habitat loss and CWD. The core areas identified in this paper provide focus points for any subsequent elk management. The sooner action is taken to reduce the impact of the identified threats, the greater the chance of a full recovery across elk range and the prevention of further population declines (Hutchings et al. 2012). When mitigating risk factors for a species like elk that relies heavily on populations in protected areas, direct human intervention limiting hunting and further fragmentation of these areas has the most impact (Rivard et al. 2000). In the case of prairie-parkland elk, the reduction of hunting in some areas or increased monitoring of elk in response to hunting to determine sustainable levels could reduce the negative impacts caused by hunting. Within the forestry industry, efforts could be made to reduce harvesting altogether in highly vulnerable areas and leave some vegetation cover when harvesting to create more elk friendly habitat. CWD management should focus on monitoring prevalence, while identifying strategies for isolation and elimination of disease outbreaks (Williams et al. 2002). Future research should focus on the human perspectives on elk conservation and management options, as well as quantitative estimates of elk populations and CWD prevalence rates (Needham et al. 2004; Lindenmayer & Hunter 2010).

CHAPTER 5: GENERAL DISCUSSION

5.1 Review

The purpose of my thesis research was to identify and understand elk distribution patterns in the prairie-parkland region using local ecological knowledge and biological research techniques, and determine the implications of that distribution for species persistence. As prior research on prairie-parkland elk is extremely limited to two geographic areas (Riding Mountain National Park [21 peer reviewed publications in the last ten years] and Cypress Hills [1 peer reviewed publications in the last ten years]), my findings provide a valuable initial framework for any conservation planning and decision making involving prairie-parkland elk. The objectives of my thesis were to: 1) generate spatial models of elk distribution using three different elk location datasets; (2) validate and compare the distribution models produced; (3) identify core areas of high quality elk habitat and assess the vulnerability of these areas to habitat loss, disease and population reduction; (4) clarify the role protected areas play in determining high quality elk habitat; and (5) define areas of priority conservation concern for elk. In this chapter, I will review key findings and provide recommendations regarding future priorities for elk managers.

5.2 Key Findings

5.2.1. *Chapter 3: Applying local ecological knowledge with biological research to map elk distribution in the prairie-parklands of Canada*

Biologists are increasingly recognizing that the knowledge held by local people, while different from knowledge gained by biological research, can be valuable and useful to conservation efforts particularly when studying species over a large spatial or time scales (Berkes et al. 2000a; Gagnon & Berteaux 2009). Understanding species distribution, and the factors it is dictated by, can be used to ecological and evolutionary insights (Elith & Leathwick 2009). In this paper, I applied a novel quantitative method that applied local ecological knowledge with radio-collar data to determine elk distribution. I used a semi-directed interview format to gain local knowledge in participatory mapping sessions.

Elk distribution was determined by 10 environmental predictor variables. Of those, paved road density, distance to protected area, herbaceous vegetation density, mixed wood forest density and deciduous forest density most influenced elk selection patterns. The distribution models made with radio-collar data only, local knowledge only and both datasets were all validated against an independent dataset of elk locations and were found to be accurate. None of the models generated were significantly different from each other. In addition to determining previously unknown elk distribution, my research also demonstrated that for this research question, local ecological knowledge was an equivalent and more economical alternative to radio-collar data.

My approach has great applicability to other studies determining species distribution. Predicting species distribution often depends on datasets that have limited utility because they were collected for another purpose or do not cover the entire study area and require extensive extrapolation (Loiselle et al. 2003). Employing LEK as an alternative to traditional biological research techniques may help overcome these pitfalls. My research also provided an alternative perspective regarding the use and comparison of LEK. Within the literature, much debate has focused on how to use, validate and compare LEK or TEK with technical biological data (Moller et al. 2004; Brook & McLachlan 2005; Gilchrist et al. 2005). A key challenge in the use of TEK and LEK is ensuring that the knowledge is adequately represented in the research, while also evaluating the data to ensure that it meets the standards of high quality that conventional expert-based research data is held to (Polfus et al. 2014). In my research (Chapter 3), I was able to translate both knowledge types with a common method and evaluated both using the same independent dataset. While I was interested in how LEK could be an alternative to radio-collar data, I was also concerned with identifying the strengths and weaknesses of both, recognising that no dataset is a perfect reflection of reality and that all data has some bias (Nadasdy 1999; Brook & McLachlan 2005). As such, I performed the analysis with the idea that neither LEK or radio-collar data was inherently more right when answering my research question. Overall, in this paper I was able to give both data forms the respect they deserve, while also achieving the main objective of identifying elk distribution.

5.2.2. *Chapter 4: Identifying conservation priorities for the persistence of remnant elk populations on the Canadian prairies*

Effective wildlife management relies on accurate identification of species' distribution and the ability to predict how future environmental changes may affect this distribution (Cianfrani et al. 2010). In this paper, I explored where the most valuable elk habitat is located and how threats to elk persistence overlay elk habitat. The locations of the identified core high quality habitat areas, and the areas defined as a priorities for conservation and concern, are valuable tools that can be used to better identify future research needs and focus current management initiatives (Margules & Pressey 2000; Hebblewhite et al. 2012).

Using measures of irreplaceability and vulnerability, I identified conservation priority areas (Pressey & Taffs 2001). I then assessed how connectivity between core areas may help or hinder elk population numbers and explored the role protected areas play in defining high quality elk habitat. The high quality areas of elk habitat were mainly clustered along the transition zone between the prairie and boreal forest ecosystems, with a few outliers in southern Saskatchewan and eastern Manitoba. My connectivity analysis indicates that core areas of high quality habitat are relatively isolated from other core areas in the study area, but exhibit a high degree of connectivity with each other. The core areas endemic with CWD are located within this highly connected matrix. Over 85% of high quality habitat areas are located within a protected area, which may be the result of protected areas consisting of native habitat.

This analysis was a critical first step in assessing the conservation status of prairie-parkland elk. In order to make any conservation decision, it is vital to understand the patterns of species residency on the landscape, and the factors that dictate them (Johnson et al. 2004; Higgins et al. 2012). For instance, evaluating the performance of protected areas is a vital step in a conservation plan, however in order to evaluate protected areas, distinguishing how they influence and protect current species is required (Knight et al. 2008). CWD is recognised as a large management issue for all cervid populations in Alberta, Saskatchewan and Manitoba (Conner et al. 2008; Rees et al. 2012). However, very little research regarding host distribution and movement through endemic areas has taken place. By overlaying identified corridors with disease endemic locations, I was able to provide the first insight into potential CWD risk for endemic elk populations. This paper also brought awareness to the state of elk populations and available habitat in this area since regional extirpation. By completing this analysis, I was able to demonstrate that elk populations in the prairie-parkland region have not recovered their distribution in the prairie portion of their range, and in areas where they are well established, they continue to be vulnerable to anthropogenic threats, as well as stochastic events like disease.

5.3 Recommendations

5.3.1 Stakeholder engagement

Conservation science is being increasingly recognized as a field influenced by human perceptions and values (Berkes 2004; Lindenmayer & Hunter 2010). Researchers often perform experiments or suggest management plans in a vacuum separate from human dimensions, which can result in mediocre conservation outcomes (Riley et al. 2002). Management plans created in isolation from the impacted communities are less effective because implementation is often crippled by a lack of stakeholder support, buy in or compliance (Moller et al. 2004). This can result in limited resources being spent on polices that will not effectively reach their goal due to stakeholder dis-engagement (Bottrill et al. 2008). Past research has shown that researchers who include stakeholders in the research and policy creation process have better long-term success in achieving conservation goals (Garmendia & Stagl 2010). Developing policies with stakeholders can result in policies and outcomes that are more favourable for local people, and therefore much more successful and sustainable in the long term (Reed 2008; Danielsen et al. 2009).

One way to engage communities and stakeholders is to use local ecological knowledge as a research technique (Reed 2008; Danielsen et al. 2009; Martin et al. 2012). While my project did not explicitly focus on stakeholder engagement, I hoped to generate discussion regarding elk management issues in the participatory mapping and workshop process. In the study area, there are many different elk users with different, sometimes confounding priorities. Within knowledge gathering sessions, I talked with biologists, conservation officers, First Nations hunter, farmers and wildlife viewers. Each of these groups has a different opinion regarding how elk should be managed and has a different knowledge base or type of knowledge about elk (Houde 2007). For example, some participants clearly understood the mechanics of CWD transmission and the implications of increased prevalence, while others had no knowledge of the disease or which species were infected by it. Perspectives and attitudes towards elk tend to be dictated by individual's interactions with them (Leuschner et al. 1989; Heydlauff et al. 2006; Crank et al. 2010). A farmer whose crops are damaged by elk every year may view elk presence less favourably than a recreationalist who visits parks to photograph them (Hegel et al. 2009). Any management policy needs to include the variety of perspectives held by those most influenced by elk. There are many potential ways to engage stakeholders: gather LEK to answer additional research questions; understand, document and discuss the varying perspectives regarding elk in the region; identify potential management strategies and how they would be perceived; determine stakeholder concerns about elk; recruit individuals for management efforts, for instance targeted hunting in CWD areas or reduction of baiting and feeding cervids in CWD endemic zones (Saunders et al. 2012; Sorensen et al. 2014); and discuss matters of interest to public health like the zoonotic potential of CWD or bovine tuberculosis. The concept of One Health recognizes that human health, animal health and environmental health are inextricably linked (One Health Initiative 2014). One Health is multi-disciplinary approach to ecological problems that recognizes the impact of environmental health on socio-ecological systems (Zinsstag et al. 2011). A prime opportunity to apply the concept of One Health would be stakeholder engagement and knowledge sharing with researchers and managers in this region.

5.3.2. *CWD management*

CWD has the potential to greatly impact prairie-parkland elk through population reduction as well as socio-economic factors (Bollinger et al. 2004; Sigurdson 2008). Although current CWD prevalence of elk is estimated to be 0.26% in endemic areas, the experience gained in CWD outbreaks in other areas demonstrate how quickly prevalence can increase and once it does, how difficult the disease is to manage (Saunders et al. 2012; Bollinger et al. 2014). Once the disease is established, it is virtually impossible to eradicate due to asymptomatic infected animals, environmental prion transmission, lack of an easily administered test and no prophylactic (Sigurdson 2008; Tapscott 2011; Gilch et al. 2011) which will result in severe population reduction if it becomes established (Gross & Miller 2001). Management options for CWD are limited and typically focus on limiting transmission through population reduction. Reducing the transportation or reintroduction of host species, banning baiting and feeding of cervids, as well as close monitoring of captive farming operation are other options with less of an impact (Williams et al. 2002; Conner et al. 2008; Holsman et al. 2010; Sorensen et al. 2014). Illinois was able to successfully reduce CWD prevalence using continuous, frequent culls of moderate intensity and by starting a management program shortly after the identification of CWD (Mateus-Pinilla et al. 2013). In contrast, Wisconsin had greater difficulty managing CWD due to lack of public support and compliance in culling initiatives which resulted in a higher prevalence, even though Wisconsin also implemented its management program shortly after the discovery of CWD (Holsman et al. 2010).

Current policy regarding CWD in Saskatchewan is limited and does not require hunters to submit heads for testing. Hunters were required to submit heads in endemic areas beginning in 1997, but this program was cancelled in 2013 (Rees et al. 2012; Canadian Cooperative Wildlife Health Centre 2013a). Manitoba sporadically tests for the disease in animals from areas that are close to the Saskatchewan but does not have a consistent program. In order to prevent a potentially catastrophic disease situation for cervids in Saskatchewan, surveillance and management of CWD is necessary. Although it may be too late to reduce prevalence in endemic areas, management and intervention could prevent the spread and infection of other areas and populations (Cotter 2013).

5.3.3. *Additional research*

Thus far, research involving prairie-parkland elk is spatially limited to specific areas, which has resulted in the majority of elk populations being minimally or never studied. The lack of data regarding most of the populations in the study system makes to it difficult to accurately manage elk and perform further quantitative assessments, such as population viability analyses or epidemiological modelling. Information that is a key determinant in a variety of factors: population sizes in core areas, population demographics, actual movement routes used by elk, and genetic relatedness of sub-populations across the study area, is unknown. Without these measures, it is very difficult to develop the quantitative predictive models that determine future disease spread through populations and project the population trajectory in response to both stochastic and deterministic extinction events. For instance, identifying genetic relatedness between populations can provide insight into dispersal routes and evolutionary responses to the local environment (Wang et al. 2009). Previous research on the genetic diversity of North American elk identified that elk in Riding Mountain National Park were genetically different from Elk Island National Park in Alberta, which is also in the prairie-parkland range (Polziehn et al. 2000). Identifying where this difference occurs is critical to understanding how genetic relatedness may prevent the spread of CWD and correct any assumptions made in the connectivity analysis. Overall, in order to create a management plan achieves the purpose of maintaining or increasing the population of prairie-parkland elk, more research is required to answer critical questions.

5.4 Concluding Remarks

Managing a large herbivore prone to human-wildlife conflict on an agricultural dominated landscape is a complex and value-laden task (Messmer 2000; Gordon 2009). To make the best possible decisions, it is necessary to have a detailed understanding of that species' spatial distribution, and the factors that influence its presence and survival on the landscape (Fischer & Lindenmayer 2007; Austin 2007). My research indicates that elk populations remain closely tied to the forest fringe and that protected areas of all types play a large role in current elk population persistence. Very little high quality elk habitat exists on the agriculture-dominated prairie.

My research also demonstrated that local experts in the prairie-parkland region are knowledgeable about the habits of the elk they interact with. Local people may be able to provide additional data regarding elk behaviour in the study area, which may be useful given the lack of information currently available. Using local knowledge in a quantitative framework can provide valuable information in a conservation situation where resources and data are limited (Bohensky & Maru 2011).

This thesis highlights the threats that currently face elk population persistence and expansion in the Canadian prairie-parkland region. Disease and habitat loss have the potential to reduce elk populations through cumulative negative effects. In order to mitigate these threats and ensure the presence of elk on the prairie landscape, management programs that involve local people, monitor and prevent disease spread, and reduce habitat loss in the areas most important to elk must be created. Elk are a resilient species with the capacity to adapt to widely different environments. By recognising and reducing threats to population persistence now, managers can ensure that elk remain on the prairie landscape for many years to come.

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APPENDIX

Figure A.1 – The author leading a local ecological knowledge gathering workshop in June 2011, in Tisdale, Saskatchewan.

Figure A.2 – Survey on elk habitat characteristics given to participants in local knowledge gathering sessions. The characteristics were derived from previous workshop participant responses.

How important are these characteristics to elk habitat?

Characteristic	**Circle** the face that reflects how the characteristic affects elk. ← → Good for elk / Doesn't make a difference / Bad for elk	**Score** the importance of each characteristic. 1=not important 10=very important	**Rank** the relative importance of each characteristic in order. Place the stickers provided in the table, 1=most important, 10=least important.
Close access to preferred agricultural crops			
Easy access to forest cover			
Few predators (wolves and bears)			
Highly varied topography (lowlands and hills)			
High road density			
High hunting pressure			
Open land to graze on			
Parks and other areas protected from hunting			
Seasonal movement routes or patterns in the area			
Close to rivers and wetlands			

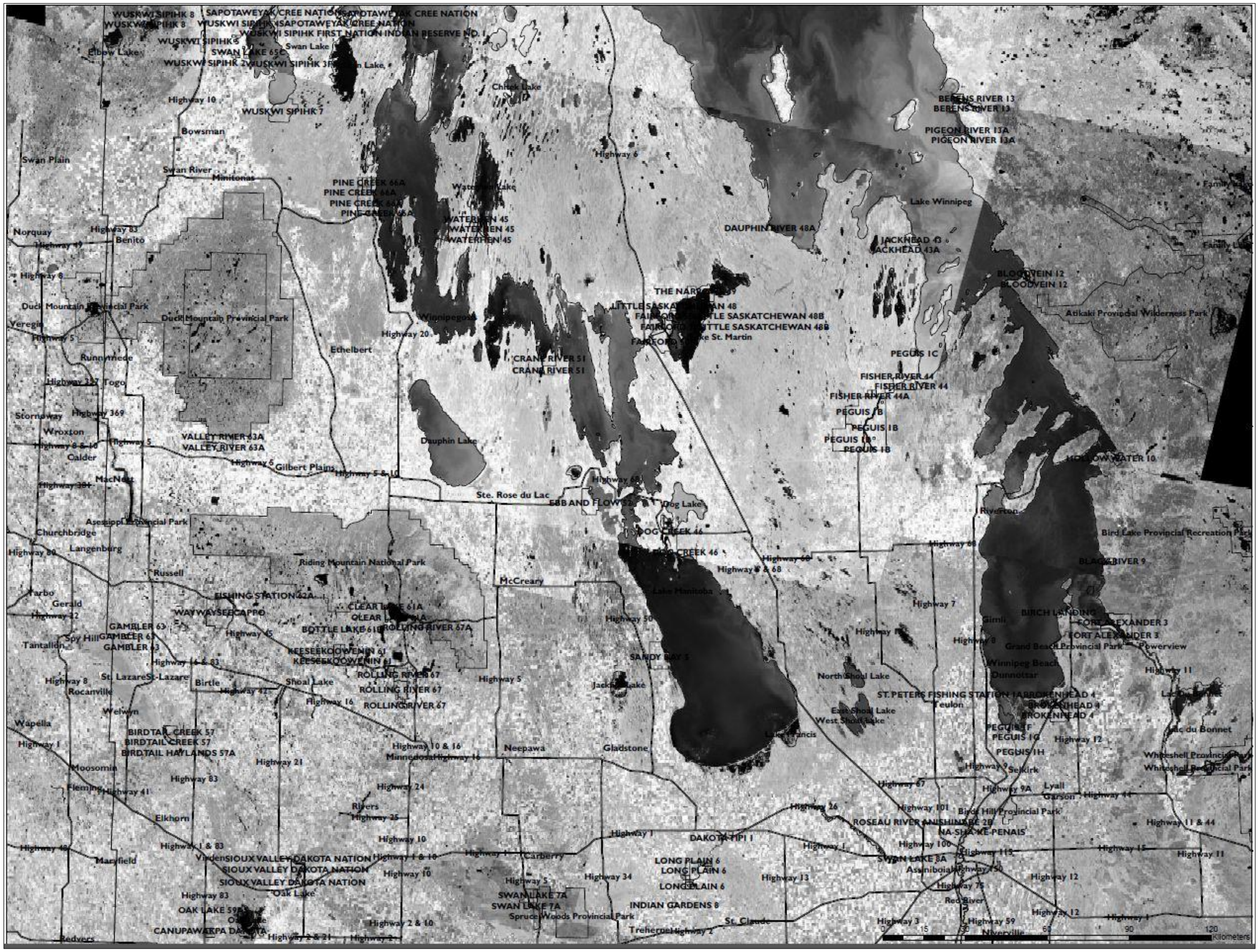

Figure A.3 – One of two maps of Manitoba used in the Grandview workshop to document local ecological knowledge. Five maps also using satellite images were created for Saskatchewan.

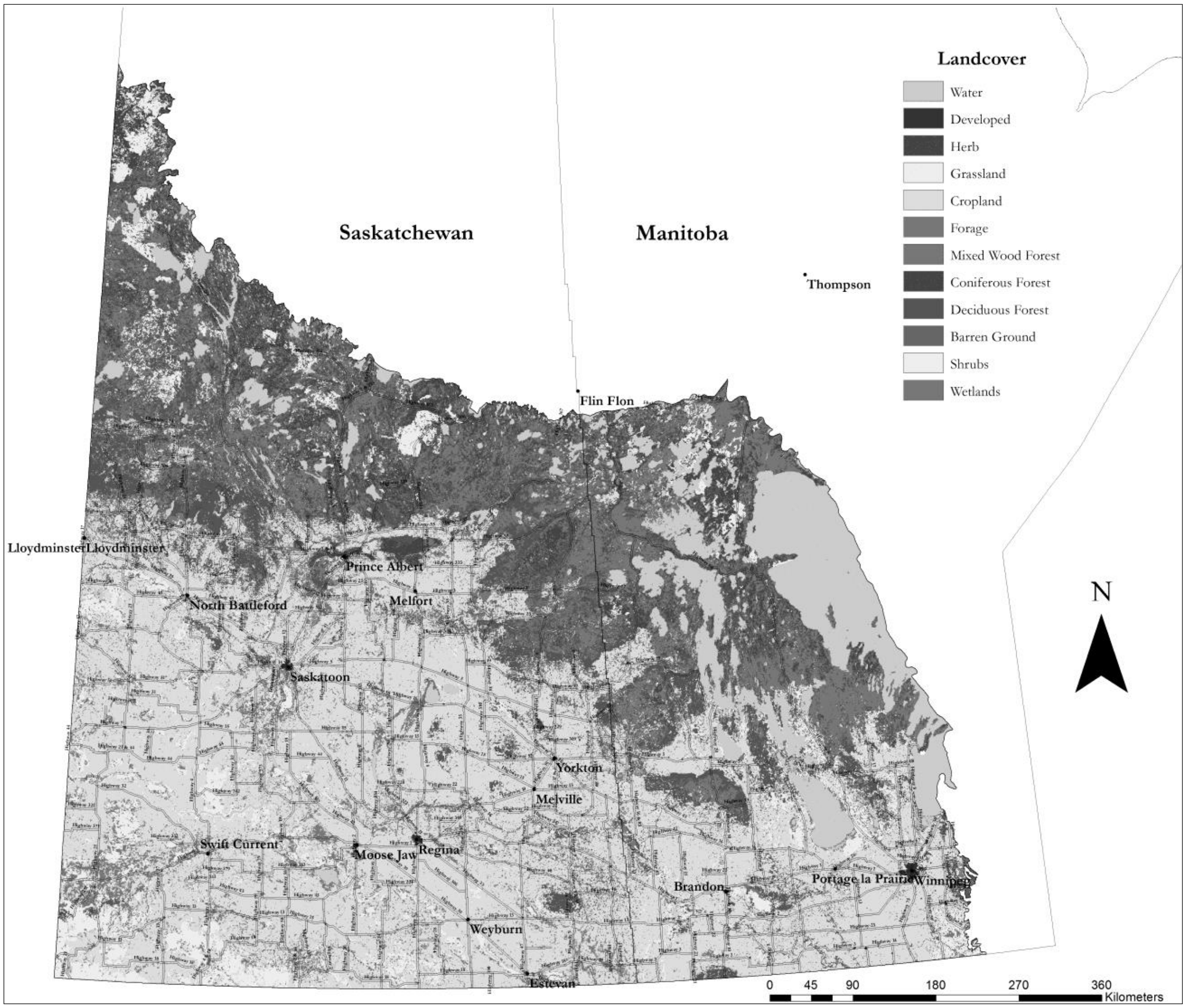

Figure A.4 – Another version of the maps used in the participatory mapping exercise to document local ecological knowledge. This map was developed using the land cover of the entire study area.

Table A.1 – Summary of primary group affiliation of local knowledge gathering session participants. Many participants may belong to more than one group, but are listed here in the group that they are predominantly affiliated with.

Primary Group Membership	# of Participants
Conservation Officer	33
Aboriginal Hunter	16
Farmer	7
Biologist	6
Hunter	9